REVIEW DRAFT

TECHNICAL SUPPORT DOCUMENT

EVALUATION OF THE POTENTIAL FOR RECYCLING OF SCRAP METALS FROM NUCLEAR FACILITIES

VOLUME 1 OF 3: CHAPTERS 1-10

Prepared by:

S. Cohen & Associates, Inc.
1355 Beverly Road
McLean, Virginia 22101

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U.S. Environmental Protection Agency
Office of Radiation and Indoor Air
401 M Street, S.W.
Washington, D.C. 20460

Martin Offutt
Work Assignment Manager

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EXECUTIVE SUMMARY

The operation, decommissioning, and cleanup of nuclear facilities owned by the Federal government and licensed by the U.S. Nuclear Regulatory Commission (NRC) will likely generate large quantities of scrap metal. Some of this metal may be moderately contaminated with radioactivity as a result of deposition or neutron activation. Current practice is to dispose of such material in a licensed, low-level waste disposal facility. The U.S. Environmental Protection Agency (EPA) is evaluating the potential for recycling scrap metal from nuclear facilities as an alternative disposition option. The Agency is also assessing the need for regulatory action to ensure that the recycle of this scrap metal does not endanger public health and safety.

This Technical Support Document (TSD) summarizes the technical information used by EPA in its evaluation. In a separate document, "Radiation Protection Standards for Scrap Metal: Preliminary Cost-Benefit Analysis" (IEC 97), EPA describes its analysis of the potential costs and benefits of recycling scrap metal from nuclear facilities.

The primary objectives of the Agency's technical analysis were to:

1. characterize the potential sources of scrap metal, including uncertainties, that may be available for recycling;
2. estimate, including uncertainties, the potential normalized annual dose and normalized lifetime risk to the reasonably maximally exposed individual (RMEI) associated with the release of scrap metal from nuclear facilities;
3. estimate, including uncertainties, the potential normalized collective dose and normalized collective risk to the exposed population associated with the release of scrap metal; and
4. estimate, including uncertainties, the minimum detectable concentration (MDC) of radionuclides contained within or on the surface of scrap metal.

ES.1 CHARACTERIZATION OF THE POTENTIAL SOURCES OF SCRAP METAL FROM NUCLEAR FACILITIES

The principal administrative authorities responsible for the management of scrap metal from nuclear facilities are the U.S. Department of Energy (DOE), NRC, the Department of Defense (DoD), and State or Superfund authorities. The nuclear facilities managed by the DOE and commercial nuclear power plants licensed by NRC are the largest potential sources of scrap metal from nuclear facilities in the United States. The DOE facilities and NRC-licensed commercial power reactors were the basis for the evaluations reported in the TSD.

ES.1.1 DOE Scrap Metal

Information in several reports (DOE 95; DOE 96; EPA 95) was used to estimate the inventory of scrap metal currently in storage at DOE facilities. Based on these reports, the current inventory of scrap metal potentially available for recycle is about 171,000 metric tonnes. However, DOE 96 cautions that its efforts were to

> "not attempt to capture the exact amount of each material in inventory. Rather, [they attempt] to capture the general magnitude of the inventory of each material (MIN 96)."

With respect to potential future sources of DOE scrap metal, DOE's Office of Environmental Restoration Decommissioning Inventory slated 865 structures for future decommissioning (as of June 1995). Future DOE scrap metal quantities will be closely linked to projected decommissioning activities at DOE sites that make up the nuclear weapons complex. At some sites, virtually all structures and their contents will be dismantled and removed; at other sites, decommissioning may be limited, and the DOE will continue selected operations considered crucial to national security or important to national research. To date, final decisions and commitments for decommissioning all of its facilities have not yet been made.

Based on available data, the quantity of future sources of DOE scrap metal is estimated to be about 925,000 metric tonnes. Table ES-1 provides summary estimates of the combined quantity of existing and future scrap metal from DOE facilities. Of the more than one million metric tonnes of scrap, about 85 percent represents carbon steel with nearly equal quantities of copper, nickel, aluminum, and stainless steel representing the remainder. These values may substantially underestimate the total scrap metal quantities because current plans for future decommissioning of DOE sites have not yet been finalized.

Table ES-1. Summary Data for Existing and Future Contaminated Scrap at DOE Facilities* (metric tonnes)

Site Name	Scrap Metal Database Volume	Metal Type													
		Al	C. Steel	S. Steel	Copper	Nickel	Monel	Lead	Cast Iron	Black Iron	Graphite	Cu/Brass	Tin/Fe	Other	Misc.
Fernald	139,841	—	101,753	—	38,088	—	—	—	—	—	—	—	—	—	—
Hanford	92,175	684	87,020	787	—	24	—	291	—	—	1,632	5	1	1,711	20
Idaho	34,213	30	19,195	14,733	44	44	—	110	4	7	—	8	2	36	—
LANL	5,785	40	5,568	177	—	—	—	—	—	—	—	—	—	—	—
NTS	264	11	204	15	—	17	—	—	—	—	—	2	1	—	14
ORNL	1,129	18	992	117	2	—	—	—	—	—	—	—	—	—	—
Y-12	9,065	33	8,392	602	38	—	—	—	—	—	—	—	—	—	—
K-25	242,063	7,988	232,953	753	304	—	65	—	—	—	—	—	—	—	—
Paducah	279,260	21,161	212,917	190	198	44,794	—	—	—	—	—	—	—	—	—
Portsmouth	197,986	6,130	191,412	18	408	—	18	—	—	—	—	—	—	—	—
Rocky Flats	50,846	—	33,666	2,454	14,726	—	—	—	—	—	—	—	—	—	—
SRS	16,237	14	10,403	5,809	11	—	—	—	—	—	—	—	—	—	—
Weldon Sp	27,839	510	26,877	406	46	—	—	—	—	—	—	—	—	—	—
TOTAL	1,096,703	36,619	931,352	26,061	53,865	44,879	83	401	4	7	1,632	15	4	1,747	34
Percent of Total	100.00	3.34	84.92	2.38	4.91	4.09	0.01	0.04	3.6E-6	6.4E-6	0.15	1.4E-5	3.6E-6	0.16	3.1E-5

* Includes all metals which may be available for recycle.

ES.1.2 Commercial Nuclear Power Plants

The U.S. commercial nuclear power industry includes 123 reactor plants. At present, eight of these reactors have been shutdown; in the next two to three decades, most of the reactors currently in operation will have reached their projected forty-year lifetime. A great deal of information and data has been compiled by the NRC and the individual utilities pertinent to the decommissioning of these facilities and the associated quantities and characteristics of the scrap metal that will be produced.

In the 1976-1980 time frame, two studies were done for the NRC which examined the technology, safety, and costs of decommissioning large reference nuclear power plants. These studies reflected the industrial and regulatory situation of the time. To support the final Decommissioning Rule issued by the NRC in 1988, the earlier studies were updated with two additional topical reports. These four reports, along with several other NRC reports and selected decommissioning plans on file with the Commission, were the primary sources of information used by EPA to characterize Reference Pressurized Water Reactor (PWR) and Boiling Water Reactor (BWR) facilities and to derive estimates of scrap metal inventories for the industry at large.

Quantities of both carbon steel and stainless steel will potentially be available for recycling from decommissioned commercial nuclear power plants. Estimates for the entire commercial nuclear industry were derived by taking Reference BWR and Reference PWR values and applying plant-specific scaling factors for each of the 40 BWRs and 83 PWRs in existence. Approximately 600,000 metric tonnes of steel may become available over time for recycling. About 80% of this metal is carbon steel with stainless steel representing most of the balance.

ES.2 ESTIMATES OF THE NORMALIZED ANNUAL DOSE AND NORMALIZED LIFETIME RISK

In order to evaluate the potential impacts of recycling scrap metal from nuclear facilities, a relationship must be established between the possible levels of radioactivity in the scrap and the potential doses and risks to individuals that may be exposed to the scrap or to the products and byproducts of the recycling process. For the purposes of establishing this relationship, EPA identified over 60 different categories of individuals that have the potential to receive some level of exposure to residual radioactivity contained in the scrap. Within each of these categories, there will be a range of exposure levels depending on the actual activities of each individual. As part of its analysis, the Agency determined the limiting category for each of the radionuclides considered. (Limiting refers to that category which has the highest potential doses associated with a given radionuclide). The Agency also determined, within each category, the individual(s) who has the potential to receive high end exposures, e.g., 90th percentile. These individuals are referred to as reasonably maximally exposed individuals (RMEI). It is unlikely that many individuals within or outside the group could receive exposures significantly greater than those received by the RMEI; most individuals that may be exposed are likely to receive exposures that are substantially lower than those received by the RMEI.

Since doses and risks to individuals are directly proportional to the residual radioactivity in the scrap, the Agency has chosen to express this relationship in terms of a "normalized dose" for each radionuclide of concern. The normalized dose is expressed in units of millirem per year (mrem/y) effective dose equivalent (EDE) per picoCurie per gram (pCi/g) of specific radionuclides in released scrap metal.

The normalized dose is a useful metric because, for any free release criterion established in units of mrem/y, the normalized dose can be used to derive the average[1] radionuclide concentration level in scrap metal that corresponds to the criterion. If the average concentration of a given radionuclide in scrap metal is known, the annual dose to the RMEI resulting from its release can

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[1] When deriving the normalized annual dose to the RMEI, the volume of scrap metal over which the radionuclide concentrations are averaged differs depending on the exposed individual. For example, when deriving the normalized annual dose to a mill worker, the volume of scrap metal of concern is the entire scrap metal throughput at the mill over a year. However, when deriving the normalized annual dose to a user of a product made from scrap metal from a nuclear facility, the averaging volume is the volume of scrap metal required to make the product.

be estimated by multiplying the normalized dose by the average concentration of the radionuclide in the scrap metal.

The normalized risk is similar to the normalized dose except that it is expressed in units of the lifetime risk of cancer associated with one year's exposure resulting from recycling operations, per pCi/g of a given radionuclide in scrap metal. It can be used to derive that concentration of a given radionuclide in scrap metal that corresponds to a given risk. It can also be used to derive the potential lifetime risk of cancer for the RMEI from one year's exposure associated with the release of scrap metal containing a known average concentration of a given radionuclide.

In order to derive the normalized individual doses, visits were made to scrap yards and steel mills to gather information on the handling and processing of scrap metal for recycling and on the disposition of the products and byproducts associated with the overall recycling process for carbon steel. Table ES-2 presents the results of these analyses. Six of the over 60 categories of individuals evaluated were found to be limiting. A particular category is considered limiting for a particular radionuclide because of factors concerning how the radionuclide partitions during melting; whether the radionuclide is diluted; whether the radionuclide has the potential to escape the baghouse filter; the mobility and bioavailability of the radionuclide in the environment; and the activities and living habits of the individuals that may come into contact with the scrap, steel, slag, baghouse dust, or airborne effluents from the mill.

Table ES-2. Derived Normalized Doses and Risks to the RMEI from One Year of Exposure

Nuclide	Limiting Scenario	Dose (mrem per pCi/g)	Lifetime Risk of Cancer (per pCi/g)
C-14	Airborne effluent emissions	8.66E-04	4.28E-10
Mn-54	Lathe operator	2.02E-01	1.54E-07
Fe-55	Cutting scrap	6.69E-06	2.71E-12
Co-60	Lathe operator	8.99E-01	6.84E-07
Ni-59	Cutting scrap	4.39E-06	1.55E-12
Ni-63	Cutting scrap	1.07E-05	4.41E-12
Zn-65	Cutting scrap	9.61E-02	7.31E-08
Sr-90+D	Slag leachate in groundwater	1.52E+00	5.51E-07
Nb-94	Slag pile worker	4.74E-01	3.60E-07
Mo-93	Cutting scrap	5.65E-05	1.17E-11
Tc-99	Cutting scrap	2.15E-05	1.41E-11
Ru-106+D	Lathe operator	5.16E-02	3.93E-08
Ag-110m+D	Lathe operator	6.29E-01	4.78E-07
Sb-125+D	Cutting scrap	6.37E-02	4.85E-08
I-129	Airborne effluent emissions	7.91E-01	5.04E-07
Cs-134	Cutting scrap	2.46E-01	1.87E-07
Cs-137+D	Cutting scrap	8.91E-02	6.77E-08
Ce-144+D	Slag pile worker	1.77E-02	1.36E-08
Pm-147	Slag pile worker	1.42E-04	8.31E-11
Eu-152	Slag pile worker	3.44E-01	2.61E-07
Pb-210+D	EAF furnace operator	3.08E+00	4.37E-07
Ra-226+D	Slag pile worker	6.27E-01	4.36E-07
Ra-228+D	Slag pile worker	3.68E-01	2.36E-07
Ac-227+D	Cutting scrap	8.00E+00	1.35E-07
Th-228+D	Slag pile worker	1.35E+00	6.17E-06
Th-229+D	Slag pile worker	4.37E+00	2.32E-07
Th-230	Slag pile worker	6.42E-01	3.44E-08
Th-232	Slag pile worker	2.84E+00	3.34E-08
Pa-231	Slag pile worker	2.51E+00	5.20E-08
U-234	Slag pile worker	3.14E-01	3.31E-08
U-235+D	Slag pile worker	3.28E-01	5.90E-08
U-238+D	Slag pile worker	2.89E-01	3.55E-08
Np-237+D	Slag pile worker	1.53E+00	1.36E-07

Table ES-2. Derived Normalized Doses and Risks to the RMEI from One Year of Exposure (Continued)

Nuclide	Limiting Scenario	Dose (mrem per pCi/g)	Lifetime Risk of Cancer (per pCi/g)
C-14	Airborne effluent emissions	8.66E-04	4.28E-10
Pu-238	Slag pile worker	6.82E-01	4.78E-08
Pu-239	Slag pile worker	7.29E-01	4.73E-08
Pu-240	Slag pile worker	7.29E-01	4.73E-08
Pu-241+D	Slag pile worker	1.17E-02	4.01E-10
Pu-242	Slag pile worker	6.93E-01	4.46E-08
Am-241	Slag pile worker	1.21E+00	1.07E-07
Cm-244	Slag pile worker	6.75E-01	6.69E-08
U-Series	EAF furnace operator	3.61E+00	4.78E-07
U-Separ.	Slag pile worker	6.18E-01	7.14E-08
U-Deplete	Slag pile worker	3.22E-01	3.95E-08
Th-Series	Slag pile worker	4.55E+00	8.86E-07

A semi-quantitative uncertainty analysis was performed which evaluated the uncertainty/variability in the dose evaluation results due to uncertainty/variability in the calculational parameters and assumptions. In addition, sensitivity analyses were performed to evaluate which parameters contributed most to the uncertainty. Table ES-3 provides the results of the semi-quantitative uncertainty/sensitivity analysis.

In Table ES-3, the critical population group is identified for each radionuclide, along with the dominant exposure pathway(s). An "upper-end multiplier" and "lower-end divisor" is provided. The upper-end multiplier and lower-end divisor define the potential range (and therefore the uncertainty) of the normalized RMEI dose. The values for the multipliers and divisors are largely based on professional judgment and are designed to bracket estimated uncertainties and variabilities for normalized RMEI doses. In general, the analyses demonstrate that the normalized doses for the RMEI could, in theory, be higher by a factor of 5 to 50, or lower by up to a factor of 100 to 500, depending on the radionuclide. The uncertainty in the normalized risks are similar.

Table ES-3. Uncertainty/Variability in Normalized Individual Dose

Radionuclides	Critical Population Group	Primary Exposure Pathway	Upper End Multiplier	Lower End Divisor	Bases/Controlling Parameters
Zn-65* Sb-125 Cs-134* Cs-137*	Scrap yard workers	External exposure	10	100	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30 fold), reduced occupancy and increased distance (3).
Ni-59/63 Mo-93 Tc-99 Ac-227+D		Inhalation	10	500	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30 fold), reduced occupancy (2), and reduced dust loading (10)
Fe-55		Soot ingestion	10	500	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30), reduced occupancy (2), and reduced soot ingestion (10)
Mn-54 Co-60 Ru-106 Ag-110m+D	Users of metal products	External exposure	5	100	Upper end due to increase in size of component and occupancy time (5). Lower end due to application of a dilution factor (30) and lower occupancy time and smaller size component (3).
Nb-94 Ce-144+D Eu-152 Ra-226+D/228+D Th-228+D	Slag pile	External exposure	40	100	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (4). Lower end due to additional dilution (30) and smaller contaminated area and occupancy time (3).
Pm-147 Th-229/230/232 Pa-231 U-234/235/238 Np-237 Pu-all Am-241 Cm-244	Slag pile workers	Inhalation	20	500	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (2). Lower end due to additional dilution (30), lower dust loading (10), and lower occupancy time (2).
Pb-210	Mill workers	Ingestion	20	500	Upper end due to elimination of dilution factor (8) and increased occupancy time and slag partition (2). Lower end due to additional dilution (30), lower soot ingestion (10), and lower occupancy time (2).

Table ES-3. Uncertainty/Variability in Normalized Individual Dose (Continued)

Radionuclides	Critical Population Group	Primary Exposure Pathway	Upper End Multiplier	Lower End Divisor	Bases/Controlling Parameters
C-14	Offsite residents - Offsite exposure to airborne emissions	Ingestion of food	50	NA†	Upper end due to elimination of dilution factor (8), closer location (3), increased intake of crops (2).
I-129				100	Lower end due to additional dilution (30), further distance (2), less intake (2).
Sr-90	Offsite residents - Ground water contaminated by slag leachate	Ground water ingestion	50	NA†	Upper end due to less dilution in ground water.
					Lower end due to elimination of ground water due to increased transit time, and soot ingestion becomes the limiting pathway.

* These radionuclides partition to baghouse dust. If it is plausible for individuals to be exposed to reconcentrated stages of the metal recovery process for prolonged periods of time, the upper end multiplier for these radionuclides could be as high as a factor of 100.

† A lower limit for these pathways is not applicable, since the lowest limiting dose will be due to a different pathway (see text).

ES.3 ESTIMATES OF THE NORMALIZED, TIME-INTEGRATED COLLECTIVE DOSE AND NORMALIZED, TIME-INTEGRATED POTENTIAL COLLECTIVE PUBLIC HEALTH IMPACTS

The normalized, time-integrated collective dose is expressed in units of the collective dose (i.e., person rem) to which a population is committed per unit activity contained in free released scrap metal. The units can be simply expressed in terms of person rem per Curie, for example. The normalized collective dose represents the sum of all individual exposures for the entire exposed population for as long as the radionuclide can reasonably be assumed to result in human exposures. The normalized collective dose is a convenient metric because, once a determination is made of the total radionuclide inventory that may be present in scrap metal, the collective dose can be determined by simply multiplying the curie inventory associated with the scrap metal by the normalized collective dose for each radionuclide.

The time-integrated normalized collective risk is similar to the time-integrated normalized collective dose except that it is expressed in units of the potential numbers of adverse health effects per Curie of each radionuclide that may be contained in scrap metal. It can be used to derive the potential number of cancers that may be attributable to the release of a given quantity of scrap metal containing a known inventory of radionuclides.

Table ES-4 presents the estimated normalized time-integrated collective doses and risks for each of the 40 radionuclides considered in these evaluations. The values were derived based on models representing the fate of each of these radionuclides. The models take into consideration the wide variety of products and byproducts that could be produced from metal recycling and the populations that may be exposed.

The column in Table ES-4 labeled "Trans" refers to the collective dose associated with the transportation of scrap metal, metal products, and byproducts comprised of scrap metal from nuclear facilities. "Air" refers to the collective offsite population doses that may occur due to airborne emissions associated with steel mills that may recycle scrap metal from nuclear facilities. "Slag" refers to the population impacts associated with radionuclides in slag produced at mills that recycle scrap from nuclear facilities and used in a wide variety of applications, such as road building. "Dust" refers to the population doses associated with the radionuclides that may accumulate in baghouse dust during mill operations. "Steel" refers to the population doses associated with the radionuclides that may partition to steel which is then used in a variety of products.

The models also take into consideration the time period over which populations may be exposed. Any radionuclides in steel products made from scrap from nuclear facilities have the potential to cause exposures for as long as the products remain in the accessible environment. The values in Table ES-4 are therefore referred to as "time-integrated" collective impacts. As is the case for the normalized doses for the RMEI, the time-integrated normalized collective doses are derived to ensure that the potential collective health impacts are not underestimated; i.e., each value represents an upper end estimate given the range of uncertainties.

A semi-quantitative uncertainty analysis for collective dose was performed which evaluated the uncertainty in the results due to uncertainty in the calculational parameters and assumptions. In addition, sensitivity analyses were performed to evaluate which parameters contributed most to the uncertainty. In general, the results of the uncertainty analyses reveal that the collective doses could be higher or lower by less than a factor of two to three. The collective risks could also be marginally higher, but the possibility exists that the risks could be zero for extremely low doses and dose rates.

Table ES-4. Normalized Time-Integrated Collective Dose and Risks (per Ci—in scrap)

Nuclide	Doses (person-rem)						Cancers	
	Trans	Air	Slag	Dust	Steel	Total	Total	Fatal
C-14	0.00E+00	4.74E+01	0.00E+00	0.00E+00	2.12E-02	4.75E+01	2.34E-02	1.62-02
Mn-54	1.42E-02	2.73E-04	1.67E-01	0.00E+00	3.80E+02	3.80E+02	2.88E-01	1.93E-01
Fe-55	0.00E+00	1.61E-06	3.98E-03	1.64E-17	2.65E-04	4.24E-03	1.57E-07	1.56E-07
Co-60	6.35E-02	1.22E-03	0.00E+00	0.00E+00	1.01E+04	1.01E+04	7.66E+00	5.13E+00
Ni-59	0.00E+00	8.80E-06	0.00E+00	0.00E+00	1.33E-03	1.34E-03	1.18E-06	1.18E-06
Ni-63	0.00E+00	4.86E-05	0.00E+00	0.00E+00	6.84E-03	6.89E-03	6.60E-06	6.58E-06
Zn-65	5.31E-03	2.44E-02	0.00E+00	1.65E+02	6.13E+01	2.26E+02	1.72E-01	1.15E-01
Sr-90+D	0.00E+00	6.40E-02	5.87E+01	3.09E+00	0.00E+00	6.18E+01	2.25E-02	1.78E-02
Nb-94	5.77E-03	8.02E-02	5.35E+02	2.81E+01	0.00E+00	5.63E+02	3.81E-01	2.54E-01
Mo-93	5.60E-07	2.71E-04	0.00E+00	0.00E+00	2.53E+01	2.53E+01	1.92E-02	1.29E-02
Tc-99	1.45E-09	1.18E-03	0.00E+00	0.00E+00	1.91E-01	1.92E-01	1.58E-04	1.24E-04
Ru-106+D	4.63E-03	1.47E-04	0.00E+00	0.00E+00	1.76E+02	1.76E+02	1.34E-01	9.01E-02
Ag-110m	6.58E-02	3.53E-04	0.00E+00	0.00E+00	1.49E+03	1.49E+03	1.13E+00	7.61E-01
Sb-125	8.78E-03	3.41E-05	0.00E+00	0.00E+00	4.02E+01	4.02E+01	3.05E-02	2.05E-02
I-129	8.19E-07	1.43E+03	0.00E+00	0.00E+00	0.00E+00	1.43E+03	9.39E-01	9.64E-02
Cs-134	1.64E-02	6.00E-02	1.33E-01	0.00E+00	0.00E+00	2.09E-01	1.29E-04	8.68E-05
Cs-137+D	5.82E-03	1.79E-01	7.42E-01	0.00E+00	0.00E+00	9.27E-01	6.39E-04	4.27E-04
Ce-144+D	1.29E-04	5.07E-04	5.04E-02	2.65E-03	0.00E+00	5.36E-02	2.73E-05	1.74E-05
Pm-147	3.27E-09	4.41E-05	6.49E-01	3.42E-02	0.00E+00	6.83E-01	2.05E-06	1.09E-06
Eu-152	4.02E-03	4.94E-03	1.01E+01	5.34E-01	0.00E+00	1.07E+01	7.77E-03	5.17E-03
Pb-210+D	9.19E-07	4.08E+00	0.00E+00	0.00E+00	0.00E+00	4.08E+00	3.67E-04	2.90E-04

Table ES-4. Normalized Time-Integrated Collective Dose and Risks (per Ci—in scrap) (Continued)

Nuclide	Doses (person-rem)						Cancers	
	Trans	Air	Slag	Dust	Steel	Total	Total	Fatal
Ra-226+D	6.44E-03	2.60E+00	2.81E+03	1.48E+02	0.00E+00	2.96E+03	7.82E-01	5.49E-01
Ra-228+D	3.31E-03	1.49E-02	1.12E+01	5.88E-01	0.00E+00	1.18E+01	8.11E-03	5.41E-03
Ac-227+D	9.45E-04	6.86E-01	2.00E+01	1.05E+00	0.00E+00	2.17E+01	3.67E-03	2.50E-03
Th-228+D	5.60E-03	2.48E-01	2.18E+00	1.15E-01	0.00E+00	2.55E+00	1.73E-03	1.16E-03
Th-229+D	7.07E-04	8.30E-01	1.99E+02	1.05E+01	0.00E+00	2.11E+02	8.09E-02	5.47E-02
Th-230	2.57E-07	9.42E-01	7.49E+02	3.94E+01	0.00E+00	7.89E+02	2.11E-01	1.53E-01
Th-232	8.08E-08	9.83E-01	1.34E+03	7.03E+01	0.00E+00	1.41E+03	9.18E-01	6.15E-01
Pa-231	7.04E-05	1.52E+00	9.68E+02	5.10E+01	0.00E+00	1.02E+03	2.07E-02	1.46E-02
U-234	6.35E-08	2.98E-01	1.55E+02	8.18E+00	0.00E+00	1.64E+02	2.59E-02	1.60E-02
U-235+D	2.48E-04	2.83E-01	1.82E+02	9.60E+00	0.00E+00	1.92E+02	5.35E-02	3.46E-02
U-238+D	6.19E-05	2.72E-01	1.50E+02	7.92E+00	0.00E+00	1.59E+02	4.02E-02	2.50E-02
Np-237+D	5.09E-04	5.01E+00	4.94E+03	2.60E+02	0.00E+00	5.20E+03	3.41E-01	2.78E-01
Pu-238	1.18E-08	3.25E-01	2.55E+00	1.34E-01	0.00E+00	3.01E+00	2.23E-04	1.97E-04
Pu-239	4.78E-08	3.65E-01	3.34E+01	1.76E+00	0.00E+00	3.56E+01	1.29E-03	1.12E-03
Pu-240	1.16E-08	3.64E-01	1.55E+01	8.15E-01	0.00E+00	1.67E+01	1.24E-03	1.08E-03
Pu-241+D	3.53E-09	5.43E-01	7.08E-02	3.73E-03	0.00E+00	6.17E-01	6.79E-05	5.88E-05
Pu-242	1.02E-08	3.47E-01	1.55E+01	8.13E-01	0.00E+00	1.66E+01	1.23E-03	1.07E-03
Am-241	4.66E-06	5.70E-01	3.13E+01	1.65E+00	0.00E+00	3.35E+01	3.57E-03	2.86E-03
Cm-244	1.02E-08	2.85E-01	3.02E-01	1.59E-02	0.00E+00	6.03E-01	3.37E-05	2.92E-05

ES.4 ESTIMATES OF THE MINIMUM DETECTABLE CONCENTRATION (MDC) OF RADIONUCLIDES CONTAINED WITHIN OR ON THE SURFACE OF SCRAP METAL

The minimum detectable concentration (MDC) for each radionuclide considered in the analysis must be known in order to be able to assess the feasibility and implementation costs of potential free release criteria. MDCs, which were derived for a variety of instrument types and survey techniques, were compared to the radionuclide contamination levels that correspond to potential release criteria. Tables ES-5 through ES-8 present the MDCs for alternative types of field survey instruments (alpha, beta, and gamma detectors), alternative survey techniques (using scans versus stationary counts), and using laboratory analysis of samples instead of field surveys.

Small areas of contamination can be detected for all but one radionuclide relative to a 15 mrem/y derived concentration limit (DCL)[2]. At a DCL corresponding to 1 mrem/y, detectability drops to 75%; only 25% of the radionuclides are detectable at a DCL corresponding to 0.1 mrem/y. Significant improvement is noted when surveying for distributed (large area) sources of contamination. In this case, one hundred percent of the radionuclides are detectable at DCLs corresponding to both 15 mrem/y and 1 mrem/y. At a DCL corresponding to 0.1 mrem/y, almost 70% of the radionuclides are detectable while scanning for large areas, while almost 90% could be detected using direct measurements.

The reported values represent ideal conditions, which are not always encountered in the field (for example, low background radiation levels and smooth surfaces are assumed). In addition, the results are presented for both large and small areas of surface contamination. An evaluation of how the MDCs may change under alternative conditions was performed and is presented in Table ES-9. Less than ideal conditions could lessen sensitivity for beta and alpha counting by several fold. The extent of the loss of detectability is less for scanning for large areas of contamination than for small, as well as for instances in which direct measurements are made. Surrogate methods may be useful in situations where there are multiple radionuclides present.

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[2] A DCL is the radionuclide concentration or surface contamination level in or on scrap metal that corresponds to a given annual dose.

Any assessment of volumetrically contaminated metal by standard field survey techniques is severely restricted by the limited range of alpha and beta radiation. Only those radionuclides with DCLs greater than a few hundred pCi/g can be detected reliably. However, laboratory analysis of samples of scrap steel or steel derived from the recycling of scrap metal provides significantly improved results. State-of-the-art laboratory methods are quite effective at detecting low levels of volumetric contamination, even down to a few tenths to even hundredths of a pCi/g. At levels corresponding to DCLs of 15 mrem/yr and 1 mrem/yr, 100% of the radionuclides evaluated can be detected. Even at a DCL corresponding to 0.1 mrem/yr, 85% of the radionuclides are detectable.

Table ES-5. Detectability of Radionuclides (Small Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC (dpm/100cm²)					DCL (dpm/100cm²)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	440	2,500	85	320	120,000	7,300	Yes	490	Yes	49	No
Ag-110m+D	3,300	23,000	ND	ND	120,000	19,000	Yes	1,200	No	120	No
Am-241	4,300	78,000	420	1,600	340,000	9,700	Yes	640	Yes	64	No
C-14	3,900	31,000	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	650	2,800	ND	ND	600,000	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	24,000	610,000	420	1,600	8.1E+06	17,000	Yes	1,200	Yes	120	No
Co-60	2,300	13,000	ND	ND	200,000	13,000	Yes	870	No	87	No
Cs-134	1,800	9,600	ND	ND	160,000	48,000	Yes	3,200	Yes	320	No
Cs-137+D	1,300	6,600	ND	ND	420,000	130,000	Yes	8,800	Yes	880	No
Eu-152	2,900	15,000	ND	ND	140,000	34,000	Yes	2,300	No	230	No
Fe-55	31,000	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	3,600	30,000	ND	ND	320,000	15,000	Yes	990	No	99	No
Mn-54	3.9E+06	1.5E+07	ND	ND	360,000	58,000	No	3,900	No	390	No
Mo-93	46,000	ND	ND	ND	1.2E+06	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	1,600	8,000	ND	ND	180,000	25,000	Yes	1, 700	Yes	170	No
Ni-59	700,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	8,800	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	1,100	7,000	440	1,600	160,000	7,700	Yes	510	Yes	51	No
Pa-231	4,600	220,000	430	1,600	860,000	4,700	Yes	310	No	31	No
Pb-210+D	960	5,600	420	1,600	3.9E+06	3,800	Yes	250	No	25	No
Pm-147	3,300	22,000	ND	ND	3.5E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes

Table ES-5. Detectability of Radionuclides (Small Area) by Surface Scan* Relative to DCLs (Continued)

Radionuclide	MDC (dpm/100cm²)					DCL (dpm/100cm²)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-238	19,000	460,000	420	1,600	8.5E+06	17,000	Yes	1,200	Yes	120	No
Pu-239	37,000	750,000	420	1,600	2.2E+06	16,000	Yes	1,100	Yes	110	No
Pu-240	19,000	370,000	420	1,600	8.7E+06	16,000	Yes	1,100	Yes	110	No
Pu-241	18,000	ND	1.7E+07	6.5E+07	6.9E+06	1.0E+06	Yes	67,000	Yes	6,700	No
Pu-242	23,000	470,000	420	1,600	1.1E+07	17,000	Yes	1,100	Yes	110	No
Ra-226	77,000	390,000	420	1,600	5.0E+06	19,000	Yes	1,300	Yes	130	No
Ra-228+D	1,100	5,600	ND	ND	370,000	64,000	Yes	4,300	Yes	430	No
Ru-106+D	840	3,500	ND	ND	1.0E+06	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	2,100	13,000	ND	ND	280,000	180,000	Yes	12,000	Yes	1,200	No
Sr-90+D	550	2,300	ND	ND	8.7E+09	40,000	Yes	2,600	Yes	260	No
Tc-99	2,200	12,000	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	580	2,700	85	320	140,000	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	400	2,200	85	320	110,000	13,000	Yes	900	Yes	90	Yes
Th-230	15,000	120,000	420	1,600	1.2E+07	18,000	Yes	1,200	Yes	120	No
Th-232	17,000	160,000	420	1,600	1.5E+07	4,100	Yes	280	No	28	No
U-234	16,000	180,000	420	1,600	9.9E+06	37,000	Yes	2,500	Yes	250	No
U-235+D	1,200	10,000	430	1,600	160,000	36,000	Yes	2,400	Yes	240	No
U-238+D	680	3,000	420	1,600	1.2E+06	41,000	Yes	2,700	Yes	270	No
Zn-65	120,000	600,000	ND	ND	740,000	120,000	Yes	8,100	No	810	No

* Scan rate = 1/3 detector width per second for beta & alpha, 15 cm per second for gamma

Table ES-6. Detectability of Radionuclides (Large Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC (dpm/100cm²)					DCL (dpm/100cm²)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	160	790	18	68	270	7,300	Yes	490	Yes	49	Yes
Ag-110m+D	1,200	7,300	ND	ND	270	19,000	Yes	1,200	Yes	120	No
Am-241	1,500	25,000	90	340	770	9,700	Yes	640	Yes	64	No
C-14	1,400	9,800	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	230	890	ND	ND	1,400	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	8,400	193,000	89	330	19,000	17,000	Yes	1,200	Yes	120	Yes
Co-60	830	4,100	ND	ND	470	13,000	Yes	870	Yes	87	No
Cs-134	640	3,000	ND	ND	370	48,000	Yes	3,200	Yes	320	No
Cs-137+D	480	2,100	ND	ND	960	130,000	Yes	8,800	Yes	880	Yes
Eu-152	1,000	4,800	ND	ND	320	34,000	Yes	2,300	Yes	230	No
Fe-55	11,000	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	1,300	9,500	ND	ND	730	15,000	Yes	990	Yes	99	No
Mn-54	1.4E+06	4.8E+06	ND	ND	820	58,000	Yes	3,900	Yes	390	No
Mo-93	16,000	ND	ND	ND	2,800	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	570	2,600	ND	ND	410	25,000	Yes	1, 700	Yes	170	No
Ni-59	250,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	3,100	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	370	2,200	92	350	360	7,700	Yes	510	Yes	51	No
Pa-231	1,600	70,000	91	340	2,000	4,700	Yes	310	Yes	31	No
Pb-210+D	340	1,800	89	330	9,000	3,800	Yes	250	Yes	25	No
Pm-147	1,200	6,900	ND	ND	7.9E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes

Table ES-6. Detectability of Radionuclides (Large Area) by Surface Scan* Relative to DCLs (Continued)

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-238	6,800	150,000	89	340	20,000	17,000	Yes	1,200	Yes	120	Yes
Pu-239	13,000	240,000	89	340	51,000	16,000	Yes	1,100	Yes	110	Yes
Pu-240	6,800	120,000	89	340	20,000	16,000	Yes	1,100	Yes	110	Yes
Pu-241	6,400	ND	3.7E+06	1.4E+07	1.6E+07	1.0E+06	Yes	67,000	Yes	6,700	Yes
Pu-242	8,300	150,000	89	340	24,000	17,000	Yes	1,100	Yes	110	Yes
Ra-226	27,000	120,000	89	330	12,000	19,000	Yes	1,300	Yes	130	Yes
Ra-228+D	380	1,800	ND	ND	860	64,000	Yes	4,300	Yes	430	Yes
Ru-106+D	300	1,100	ND	ND	2,400	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	730	4,000	ND	ND	650	180,000	Yes	12,000	Yes	1,200	Yes
Sr-90+D	190	730	ND	ND	2.0E+07	40,000	Yes	2,600	Yes	260	Yes
Tc-99	800	3,900	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	210	860	18	67	310	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	140	710	18	67	250	13,000	Yes	900	Yes	90	Yes
Th-230	5,200	37,000	89	340	28,000	18,000	Yes	1,200	Yes	120	Yes
Th-232	6,000	50,000	89	340	34,000	4,100	Yes	280	Yes	28	No
U-234	5,600	57,000	89	340	23,000	37,000	Yes	2,500	Yes	250	Yes
U-235+D	410	3,200	91	340	360	36,000	Yes	2,400	Yes	240	Yes
U-238+D	240	970	89	340	2,800	41,000	Yes	2,700	Yes	270	Yes
Zn-65	42,457	190,000	ND	ND	1,700	120,000	Yes	8,100	Yes	810	No

* Meter time constant = 10 s

Table ES-7. Detectability of Radionuclides by Direct Count* Relative to DCLs

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	70	260	6	18	91	7,300	Yes	490	Yes	49	Yes
Ag-110m+D	560	2,600	ND	ND	93	19,000	Yes	1,200	Yes	120	Yes
Am-241	690	8,000	32	91	270	9,700	Yes	640	Yes	64	Yes
C-14	620	3,200	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	100	290	ND	ND	480	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	3,800	63,000	32	90	6,400	17,000	Yes	1,200	Yes	120	Yes
Co-60	370	1,300	ND	ND	160	13,000	Yes	870	Yes	87	No
Cs-134	290	990	ND	ND	130	48,000	Yes	3,200	Yes	320	Yes
Cs-137+D	210	680	ND	ND	330	130,000	Yes	8,800	Yes	880	Yes
Eu-152	460	1,600	ND	ND	110	34,000	Yes	2,300	Yes	230	Yes
Fe-55	4,900	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	580	3,100	ND	ND	250	15,000	Yes	990	Yes	99	No
Mn-54	620,000	1.6E+06	ND	ND	280	58,000	Yes	3,900	Yes	390	Yes
Mo-93	7,400	ND	ND	ND	970	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	260	830	ND	ND	140	25,000	Yes	1, 700	Yes	170	Yes
Ni-59	110,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	1,400	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	170	720	33	93	120	7,700	Yes	510	Yes	51	Yes
Pa-231	730	23,000	33	92	680	4,700	Yes	310	Yes	31	No
Pb-210+D	150	580	32	90	3,100	3,800	Yes	250	Yes	25	No
Pm-147	520	2,300	ND	ND	2.7E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes

Table ES-7. Detectability of Radionuclides by Direct Count* Relative to DCLs (Continued)

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-238	3,100	47,000	32	90	6,700	17,000	Yes	1,200	Yes	120	Yes
Pu-239	5,900	78,000	32	90	18,000	16,000	Yes	1,100	Yes	110	Yes
Pu-240	3,100	39,000	32	90	6,900	16,000	Yes	1,100	Yes	110	Yes
Pu-241	2,900	ND	1.3E+06	3.7E+06	5.4E+06	1.0E+06	Yes	67,000	Yes	6,700	Yes
Pu-242	3,700	49,000	32	90	8,300	17,000	Yes	1,100	Yes	110	Yes
Ra-226	12,000	41,000	32	90	4,000	19,000	Yes	1,300	Yes	130	Yes
Ra-228+D	170	580	ND	ND	300	64,000	Yes	4,300	Yes	430	Yes
Ru-106+D	130	360	ND	ND	830	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	330	1,300	ND	ND	220	180,000	Yes	12,000	Yes	1,200	Yes
Sr-90+D	87	240	ND	ND	6.9E+06	40,000	Yes	2,600	Yes	260	Yes
Tc-99	360	1,300	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	93	280	6	18	110	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	63	230	6	18	84	13,000	Yes	900	Yes	90	Yes
Th-230	2,300	12,000	32	90	9,700	18,000	Yes	1,200	Yes	120	Yes
Th-232	2,700	16,000	32	90	12,000	4,100	Yes	280	Yes	28	No
U-234	2,500	18,000	32	90	7,900	37,000	Yes	2,500	Yes	250	Yes
U-235+D	190	1,000	33	92	130	36,000	Yes	2,400	Yes	240	Yes
U-238+D	110	320	32	90	960	41,000	Yes	2,700	Yes	270	Yes
Zn-65	19,000	63,000	ND	ND	580	120,000	Yes	8,100	Yes	810	Yes

* Count time = 60 s

Table ES-8. MDC of Radionuclides by Laboratory Analysis Compared to DCLs

Radionuclide	MDC (pCi/g)	DCL (pCi/g) for 15 mrem/y	Detectable	DCL (pCi/g) for 1 mrem/y	Detectable	DCL (pCi/g) for 0.1 mrem/y	Detectable
Ag-110m	0.04	24	Yes	1.6	Yes	0.16	Yes
Am-241	0.05 - 0.5	5.5	Yes	0.37	Yes	0.037	No
C-14	0.2 - 37	1.0E+06	Yes	6.9E+04	Yes	6,900	Yes
Ce-144+D	0.23	310	Yes	21	Yes	2.1	Yes
Cm-244	0.05 - 0.1	9.9	Yes	0.66	Yes	0.066	Yes
Co-60	0.01 - 0.3	17	Yes	1.1	Yes	0.11	Yes
Cs-134	0.02 - 0.4	180	Yes	12	Yes	1.2	Yes
Cs-137+D	0.007 - 0.3	490	Yes	32	Yes	3.2	Yes
Eu-152	0.02 - 0.9	13	Yes	0.86	Yes	0.086	Yes
Fe-55	1 - 30	2.3E+6	Yes	1.5E+05	Yes	1.5E+04	Yes
I-129	0.4 - 2	8,400	Yes	560	Yes	56	Yes
Mn-54	0.2 - 0.3	22	Yes	1.5	Yes	0.15	Yes
Ni-63	1 - 100	1.4E+06	Yes	9.4E+04	Yes	9,400	Yes
Pb-210+D	0.1 - 5	4.1	Yes	0.27	Yes	0.027	No
Pm-147	0.5 - 5	5.5E+05	Yes	3,600	Yes	360	Yes
Pu-238	0.02 - 0.4	9.0	Yes	0.60	Yes	0.060	Yes
Pu-239	0.02 - 0.4	8.4	Yes	0.56	Yes	0.056	Yes
Pu-241	0.02 - 20	530	Yes	35	Yes	3.5	Yes
Ra-226+D	0.02 - 0.7	7.7	Yes	0.51	Yes	0.051	Yes
Ra-228+D	0.1 - 2	14	Yes	0.96	Yes	0.096	Yes
Ru-106+D	0.2 - 1	290	Yes	19	Yes	1.9	Yes
Sb-125	0.11	670	Yes	44	Yes	4.4	Yes
Sr-90+D	0.03 - 5	130	Yes	8.4	Yes	0.84	Yes
Tc-99	0.3 - 15	8.3E+05	Yes	5.5E+04	Yes	5,500	Yes
Th-228+D	0.05 - 0.4	4.1	Yes	0.28	Yes	0.028	No
Th-230	0.05 - 0.5	9.8	Yes	0.65	Yes	0.065	Yes
Th-232	0.05 - 2	2.2	Yes	0.15	Yes	0.015	No
U-234	0.05 - 0.2	20	Yes	1.3	Yes	0.13	Yes
U-235+D	0.02 - 0.3	18	Yes	1.2	Yes	0.12	Yes
U-238+D	0.02 - 0.1	22	Yes	1.4	Yes	0.14	Yes
Zn-65	0.09 - 0.6	560	Yes	37	Yes	3.7	Yes

Table ES-9. Relative Range in MDCs*

Survey Mode	Beta	Alpha	Gamma
Direct measurement	1 - 3	1 - 5	1 - 7
Scan - small area source	1 - 7	1 - 14	0.01 - 1
Scan - large area source	1 - 4	1 - 7	1 - 8

* The values are multipliers to be applied to the MDCs.

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CHAPTER 1

INTRODUCTION

1.1 PURPOSE

The Office of Radiation and Indoor Air (ORIA) of the U.S. Environmental Protection Agency (EPA) is evaluating the potential for recycling scrap metal from nuclear facilities. The clean up of sites that are contaminated with radioactive material and the decommissioning of nuclear facilities is expected to generate large amounts of scrap metal. In fact, some sites controlled by the U.S. Department of Energy (DOE) already have accumulated significant inventories of scrap metal. EPA is considering the possible impacts of recycling scrap metal from nuclear facilities as an alternative to disposing of it in a licensed, low-level radioactive waste disposal facility.

As part of its evaluation, EPA has examined current recycle practices, existing regulatory guidance governing these practices, the inventory of contaminated scrap metal potentially available for recycle, and the possible radiological impacts associated with recycling such materials. This document summarizes the technical information used by EPA in its evaluation. In a separate document, "Radiation Protection Standards for Scrap Metal: Preliminary Cost-Benefit Analysis," (IEC 97) EPA describes its analysis of the potential costs and benefits of recycling scrap metal from nuclear facilities.

Based on the information provided in this Technical Support Document (TSD), as well as the results of the cost-benefit analyses, EPA will decide whether recycling scrap metal from nuclear facilities is viable and whether additional regulatory action is necessary to ensure that release of such materials does not endanger public health and safety.

1.2 SCOPE OF THE ANALYSIS

The information presented in the TSD focuses on scrap metals that are suspected to be moderately contaminated with radioactivity as a result of deposition or neutron activation (referred to throughout this document as "scrap metal"). Consequently, scrap metal from nuclear facilities which is not contaminated, such as that generated from locations outside radiation control areas, is not considered in EPA's evaluation. Conversely, some metals may be so contaminated with radioactivity, due to their use during the life of the facility, that they can only be disposed of as a radioactive waste and are therefore ineligible for recycling. Such scrap is also excluded from this evaluation.

The principal sources of scrap metal evaluated are the DOE complex of weapons-production and research facilities and commercial nuclear power plants. Each of these sources has the potential to produce large quantities of scrap metal. Department of Defense (DoD) and industrial facilities were not explicitly addressed in the analysis because they are relatively small sources of recyclable scrap metal.

Although many types of metals (including steels, aluminum, lead, copper, nickel, and precious metals) are used in nuclear facilities and may be available for recycling, EPA's evaluation of the impacts of recycling scrap metal to date has been limited to carbon steel. Carbon steel represents the largest quantity of metal potentially available for recycle from nuclear facilities. In addition, the annual utilization of carbon steel scrap in the United States is on the order of 70 million tons, whereas the annual utilization of aluminum, stainless steel, and copper scrap is about one million tons of each type of metal. However, the TSD does provide estimates of the amount of these other metals that might be available for recycling from nuclear facilities.

EPA also recognizes that recycling other metals may be economically significant and hence desirable to the nuclear industry. Therefore, characterization of other metals and their potential for recycle is currently being conducted. In addition, the Agency is continuing to improve its database for certain factors affecting the evaluation of the impacts of recycling carbon steel.

The TSD addresses a number of issues that are important to assessing the impacts of recycling scrap metal from nuclear facilities. The following list represents the steps EPA took in its analysis of these impacts. The information resulting from each of these steps is critical to the Agency's decision making regarding the need for regulatory action concerning release of scrap metal from nuclear facilities.

- Assessment of the available information on recycling scrap metal and identification of additional information needs.
- Screening of the diverse and complex information base on scrap metal to select the scope and content of information to be used in EPA's analyses.

- Characterization of scrap quantities, levels of contamination, and options for disposition.
- Selection of the conditions for radioactive contamination to be considered in the analyses.
- Selection and characterization of scenarios for individual and collective radiation exposures in activities including scrap generation and management, steel manufacture, and use of products made from recycled metal.
- Development and use of models and computer codes to evaluate individual and collective doses and risks from radiation exposure scenarios.
- Development of data to numerically characterize parameters used in the radiation exposure scenarios.
- Development of numerical values of normalized doses for each radionuclide of concern.
- Review of methods and their limitations for detection and measurement-characterization of radioactive contamination on or in scrap metal.
- Identification of factors that affect the potential for dilution and dispersion of radionuclides to various material streams, such as metal, slag, and dust.

1.3 ORGANIZATION OF THE TSD

The TSD is organized into three volumes. The first volume, which is comprised of 10 chapters and an executive summary, closely follows the methodology used in the analysis and described in Section 1.2 above. Chapter 2 of the TSD provides an overview of scrap metal operations in the United States and the characteristics of scrap metal potentially available for recycle from nuclear facilities. Chapter 3 describes the screening procedures used by EPA to select the scope of analyses to be conducted. This chapter also discusses the limitations of these analyses. Chapter 4 describes the principal sources of scrap metal which include the DOE complex and the commercial nuclear power industry.

Chapters 5, 6, and 7 present the basis for and results of the EPA's risk assessment for individuals. Chapter 5 describes the exposure pathways associated with unrestricted recycling of scrap metal from nuclear facilities. Chapter 6 describes the calculation of the radiological impacts associated with unrestricted recycling. Chapter 7 presents and discusses the results of the dose and risk calculations for unit radionuclide concentrations and throughput.

Chapter 8 evaluates current methods of detecting and measuring radioactivity in or on scrap metal. Chapter 9 discusses the time integrated collective dose and presents the results of calculations of potential health effects for unit radionuclide throughput for unrestricted recycle. Chapter 10 concludes the TSD with a discussion and semi-quantitative analysis of the sensitivities and uncertainties associated with the Agency's evaluation of the impacts associated with recycling scrap metal from nuclear facilities.

Volumes 2 and 3 of the TSD are comprised of a number of appendices that provide more detailed information regarding the potential sources of scrap metal, the models used to derive individual and collective doses and risks, the results of the analyses, key modeling assumptions, and the uncertainties in the data and assessments found in this report.

REFERENCE

IEC 97	Industrial Economics, Inc., "Radiation Protection Standards for Scrap Metal: Preliminary Cost-Benefit Analysis," prepared for the EPA Office of Radiation and Indoor Air, under Contract No. 68-DO-0102, Work Assignment Manager Reid Harvey, 1997.

CHAPTER 2

OVERVIEW OF SCRAP METAL OPERATIONS

This chapter provides an overview of the types and quantities of scrap metal potentially available for recycle, the operations of the scrap metal industry in the United States, and current practices for recycling scrap metal from nuclear facilities. It provides a summary description of the world of scrap metal generation and utilization; any scrap metal released from nuclear facilities for recycling becomes a part of this process. This chapter offers a perspective on the data, modeling, parameter characterizations, and associated technical issues considered by EPA in its evaluation.

A comprehensive discussion of the scrap metal industry in the United States is provided in the report, "Analysis of the Potential Recycling of Department of Energy Radioactive Scrap Metal" (SCA 95). This four-volume report evaluated the potential for recycling scrap metal from the DOE complex. The discussion in this chapter presents highlights from that report which are relevant to EPA's current analysis which addresses the potential for recycling metals from a broader number of sources. This chapter also presents updated summary estimates of scrap metal availability from nuclear facilities.

2.1 CHARACTERISTICS OF SCRAP SOURCES

Scrap metal released from nuclear facilities would become part of the ongoing scrap metal industry in the United States. The characteristics of the industry, in terms of factors such as annual production, vary from metal to metal. Comprehensive descriptions of the carbon steel, stainless steel, aluminum, copper, lead, and nickel scrap metal industries are included in SCA 95. Discussions of recycling aluminum and copper scrap are presented in Appendices B and C, respectively, of this TSD.

The DOE complex and decommissioned nuclear power reactors could generate scrap metals of many types. Possibilities include carbon steel, stainless steel, galvanized iron, copper, Inconel, lead, bronze, aluminum, brass, nickel, and precious metals, such as gold and silver. As discussed in Chapter 1, EPA has limited its current evaluation to the impacts associated with recycling carbon steel, although estimates of the available inventories of these other metals were made.

Impacts of recycling scrap metal from nuclear facilities must be evaluated for specific circumstances. Potential impacts will depend on the quantities, rates, and timing of the recycling activity and on the characteristics of the scrap metal industry for the particular metals of concern. To support its analysis, EPA has developed estimates of the availability of potential sources of scrap from the DOE complex and the commercial nuclear power plants in terms of quantities available, contamination characteristics, and timing of the metal's availability for recycle.

There is at present great uncertainty in the future availability of scrap for recycle from nuclear facilities. In terms of the DOE complex, uncertainty exists as to when and how decommissioning of these facilities and equipment might occur. Uncertainty also exists in terms of the timing of commercial reactor shutdowns and the associated decommissioning activities. Reactor licenses, most of which are valid for 40 years, might be renewed and dismantlement of the reactors might not occur for as long as 60 years after shutdown. Alternatively, some reactors are being shut down and decommissioned before their present operating licenses expire. Given these uncertainties, it is possible that the generation of scrap metal that could be recycled from existing DOE and reactor sources will span the next century.

As mentioned above, EPA estimated the potential inventory of scrap metal available for recycle. A comprehensive assessment of scrap metal generated by commercial power reactors was performed and a full report on this evaluation can be found in Appendix A of the TSD. An assessment of the available information on scrap metal inventories at DOE sites was also conducted. Numerous snapshot assessments of scrap metal inventories at DOE sites have been done. These assessments are discussed in Chapter 4 of the TSD. Actual tables of data derived from them are included in IEC 97 as part of the cost-benefit analysis.

DOE recently issued a comprehensive report on materials in inventory, including scrap metal (DOE 96a). This report presents a snapshot of scrap metal inventories during approximately the summer-of-1995 period. The inventories were characterized as clean (no radiological contamination), contaminated (known radiological contamination), and "unspecified" (potentially contaminated) scrap metals. These inventories are subject to large and rapid changes as a result of ongoing operations such as the sale of clean scrap, disposal of radioactive wastes, and batch production of scrap from decommissioning of structures. This DOE report does not include projections of scrap generation and availability since plans for future cleanup and decommissioning activities are highly uncertain.

The characterization of scrap metal inventories at DOE sites is further complicated by the use of different definitions of "scrap" at various DOE sites. For example, inventory estimates reported in DOE 96a include approximately 120,000 tons of carbon steel scrap associated with the uranium enrichment facilities at the Y-12, K-25, Paducah, and Portsmouth facilities. Other DOE reports note that these same sites contain on the order of 350,000 tons of carbon steel not yet declared scrap and therefore not included in current scrap inventories. These reports do not indicate when this metal might be declared scrap and made available for recycle. Recent developments in DOE policy concerning re-industrialization suggest that facilities at these sites might be converted to new uses so that potential scrap production would be limited.

2.2 INDUSTRY PERSPECTIVES

To gain perspective on the potential significance of recycling scrap from nuclear facilities to the scrap metal industry, EPA compared the annual industry throughput with its estimates of the potential inventories of metals available for recycling from DOE sites and commercial reactors. For carbon steel, SCA 95 shows that the annual production of iron and steel from scrap is on the order of 68 million tons. Table A4-4, (in Appendix A of the TSD), indicates that the total amount of carbon steel that could be generated from the decommissioning of all commercial nuclear reactors is about four million tons. (This number is derived from the 3.6 million metric tonnes shown in Table A4-4.) On the basis of potential for radioactivity contamination under service conditions, it is estimated that about 487,000 tons of this total would be scrap carbon steel that could be available for recycle over a 50-year period.

DOE 96a indicates, in Table 2-9, that the total current inventory of contaminated and "unspecified" carbon steel at DOE sites is 116,000 tons and the HAZWRAP report (HAZ 95) provides supplemental information which brings the estimate of the existing DOE inventory to about 171,000 tons (see Table 4-3 of the TSD). As noted above, the enrichment facilities contain another 350,000 tons of carbon steel which may be declared to be scrap. The inventory of potentially available carbon steel scrap from other DOE sites is small in comparison to the enrichment facilities since most of the material generated at these other sites will be ineligible for recycle because of relatively high levels of contamination.

In summary, the upper bound of carbon steel scrap generated by DOE and commercial reactor sources and available for free release and recycle is on the order of a few million tons. This scrap would be released to the scrap metal industry over a period of decades. Therefore, the total quantity of carbon steel scrap entering the industry is small in comparison to its annual throughput of 68 million tons.

The annual scrap industry throughput for stainless steel is approximately 1.1 million tons (SCA 95). The amount of stainless steel potentially available for recycle from nuclear reactors is on the order of 122,000 tons (see Table 5A-4). DOE 96a indicates an on-hand stainless steel scrap inventory of about 7,000 tons. Similar to carbon steel, the quantities of stainless steel potentially available for recycle from nuclear facilities (about 130,000 tons) are therefore also small in comparison to the annual throughput of the industry.

Quantities of other types of scrap metals potentially available from nuclear facilities, such as aluminum and copper, are also small in comparison with the annual industry throughput. However, these comparisons do not necessarily indicate that the potential radiological impacts of recycling this material are small--as noted above, the potential radiological impacts of recycling scrap metal released from nuclear facilities must be evaluated for specific scenarios within the scrap metal industry. As discussed in Chapter 1, the TSD describes in detail the impacts of recycling carbon steel generated by nuclear facilities. EPA will continue to evaluate the consequences of recycling other types of scrap metals from these facilities.

2.3 PRINCIPAL SCRAP METAL OPERATIONS CONSIDERED

Operations involved in the generation, management, and utilization of scrap metals that could result in human radiation exposures include:

- Activities at the scrap-generation site, including demolition of structures or decommissioning of equipment, on-site management of scrap piles, characterization of contamination, decontamination of scrap, and disposal of radioactive wastes, including scrap metal that is not available for recycle.
- Transport of scrap metal to scrap processor site.
- Operations at the scrap processor site, such as shredding and preparation for melting.
- Transport of the prepared scrap to a mill.

- Operations at a mill, including management of recycled scrap, furnace operations, handling of interim products, such as ingots, bag house operations, and slag disposal.
- Manufacture, transport, and sale of finished products.
- End use of finished products, such as automobiles, cookware, and industrial equipment.

As detailed in Chapter 5, human exposure scenarios are defined and characterized for each of these operations.

There are about 160 steel mills in the United States; theoretically, any could receive and process scrap metal generated by nuclear facilities. However, in practice, these mills maintain close relationships with nearby scrap metal dealers in order to minimize transportation costs. In turn, the scrap metal dealers receive their materials from close-by sources, again to minimize transportation costs.

It is to be expected, therefore, that nuclear facility sources of scrap metal, such as DOE sites and decommissioned commercial power reactors, will send their scrap to nearby scrap dealers who in turn will respond to orders from the mills they serve. As shown in SCA 95, each of the DOE sites that could be a significant source of scrap metal has proximate mills with electric arc furnaces and, by implication, near-by scrap dealers who could receive and process the recycled scrap. Scrap dealers and mills are also located in the vicinity of most nuclear power plants.

The capacities of individual electric arc furnaces vary widely, ranging from about 15 to 225 tons of steel per heat. The steel mills in the United States have in total about 260 electric arc furnaces; virtually all of the metal charged to these furnaces is scrap.

Because of the variations in furnace capacities, the characteristics of the working relationships between scrap dealers and mills, and the make up of individual charges to a furnace, evaluating the potential radiation doses and risks must be based on specific scenario parameters. For example, it is possible that a single charge to a small furnace could be made up entirely of scrap from a nuclear facility source, in which case any residual radioactivity in the scrap metal would be undiluted. Alternatively, it is possible that a single charge to a furnace could be a mixture of scrap from a nuclear facility, and scrap from other sources. In this case, any residual radioactivity in the scrap from nuclear facilities would be diluted. This variation in the characteristics of the furnace charge becomes significant when considering the characteristics of the products made from the metal produced.

The levels of residual radioactivity in scrap that become part of the intermediate or finished products will also depend on partitioning that occurs during furnace operations. As a result of chemical phase equilibrium phenomena, radioactive and other species will distribute among the metal-melt, slag, and vapor phases associated-with melting operations. The various elements partition among the phases in different ways; therefore, the evaluations of doses and risks for various workers, such as bag house and slag pile workers, must account for these differences. Partitioning during melting of carbon steel is discussed in Appendix E of the TSD; partitioning during melting of cast iron is discussed in Appendix F.

In summary, evaluation of radiation doses and risks associated with recycle of scrap from nuclear facilities must take into account a wide range of scrap management, operations, and practices. In addition, such an evaluation must consider a range of specific conditions, such as partitioning of radionuclide species during melting and the characteristics of an individual furnace charge. These factors will affect who is exposed and to what level of radioactivity. Given the importance of these factors, it is inappropriate to base evaluations on averages--the details of specific scenarios must be examined in order to fully evaluate the impacts of recycling scrap metal from nuclear facilities.

2.4 CURRENT RECYCLE PRACTICE FOR NUCLEAR FACILITIES

Recycle of scrap metal from nuclear facilities is currently practiced on a limited and directed basis. For example, specialty metals, with low levels of contamination, that are generated at one DOE site are given new use at another site and scrap is converted into containers used for radioactive waste disposal. Despite these initiatives, the general rule of thumb for management of scrap for both the DOE complex and facilities licensed by the U.S. Nuclear Regulatory Commission (NRC) is to dispose of it in a licensed, low-level waste disposal facility. However, the interest in recycling scrap metal from the DOE complex is growing. In September 1996, DOE Assistant Secretary Alm issued a policy encouraging the recycle of scrap metal from DOE facilities. Specifically, the policy directs "the release for unrestricted use [of] any material that meets applicable criteria." The policy goes on to state that "if decontamination for release for unrestricted use is not economically feasible, then the [carbon steel] that is recycled shall be fabricated into one-time-use containers for disposal of low-level wastes...." (DOE 96b).

Current recycle practices are governed by DOE and NRC. The NRC established, in Regulatory Guide (RG) 1.86, residual surface contamination criteria for facilities being decommissioned for unrestricted release. DOE in large measure adopted the NRC criteria in its basic radiation protection standards, DOE Order 5400.5. These criteria are discussed in more detail in Chapter 8.

It is important to note that RG 1.86 does not take into consideration potential health effects associated with recycle and re-use of materials released under the criteria. The RG 1.86 criteria were based on maximum permissible concentrations for air and water listed in 10 CFR Part 20 regulations and on the assumption that licensees should not be expected to reduce surface concentrations to levels below existing environmental background levels caused by fallout from atmospheric testing of nuclear devices (NRC 94). The estimated average dose for the 24 nuclides evaluated under the RG 1.86 criteria is about 10 millirem/year (mrem/yr).

REFERENCES

DOE 96a	U.S. Department of Energy, "Taking Stock: A Look at the Opportunities and Challenges Posed by Inventories from the Cold War Era," January 1996.
DOE 96b	U.S. Department of Energy Memorandum, A. Alm to Distribution, Subject: Policy on Recycling Radioactively Contaminated Carbon Steel, September 20, 1996.
HAZ 95	"U.S. Department of Energy Scrap Metal Inventory Report for the Office of Technology Development, Office of Environmental Management," prepared by Hazardous Waste Remedial Actions Program for the Department of Energy, DOE/HWP-167, March 1995.
IEC 97	Industrial Economics, Inc., "Radiation Protection Standards for Scrap Metal: Preliminary Cost-Benefit Analysis," prepared for the EPA Office of Radiation and Indoor Air, under Contract No. 68-DO-0102, Work Assignment Manager Reid Harvey, 1997.
NRC 94	U.S. Nuclear Regulatory Commission, Secy-94-145, May 27, 1994.
SCA 95	S. Cohen & Associates, "Analysis of the Potential Recycling of Department of Energy Radioactive Scrap Metal," prepared for the EPA Office of Radiation and Indoor Air under Contract No. 68D20155, Work Assignment 3-19, Work Assignment Manager John MacKinney, August 14, 1995.

CHAPTER 3

SCREENING PROCEDURES TO DEFINE THE SCOPE OF THE ANALYSIS

In evaluating the potential for recycling scrap metal from nuclear facilities, EPA was confronted with a vast array of issues. How much scrap metal is available? What levels of radiation contamination is it likely to have? How will it be handled? Which individuals may be at risk from exposure to the scrap as it moves through the recycling process? In order to focus its evaluation, EPA had to refine its objectives and develop a method of screening potential issues and data to determine those which were most significant to the analysis.

The objectives of EPA's analysis included characterizing the different types of existing and future sources of scrap metal from nuclear facilities and the levels of radiological contamination in or on the scrap. The Agency's analysis also defines the relationship between this contamination and the radiation doses to individuals and populations that may result from the free release of the scrap metal. This chapter presents the screening methods that were used to select the parameters of EPA's evaluation, including: specific scrap metal sources, types of metals, radionuclides, exposure scenarios and pathways, and potential types of adverse biological effects that may be associated with the free release of scrap metal from nuclear facilities.

This chapter is divided into seven sections. Section 3.1 provides background information on the specific areas of inquiry and analysis contained in the TSD. This section establishes the context within which the information regarding source, type, and radionuclide composition of scrap metal is used in the TSD. Section 3.2 describes the overall screening criteria used to select the specific sources, metal types, and radionuclides explicitly characterized and analyzed in the TSD. Section 3.3 describes the wide variety of existing and potential sources of scrap metal at nuclear facilities. This section also describes the methods used to screen these sources down to a manageable number without overlooking any potentially important source. Section 3.4 describes the different types of scrap metals (e.g., carbon steel, stainless steel, galvanized iron, copper, aluminum, etc.) potentially available for recycle and the methods used to select the specific types of metals explicitly addressed in the TSD. Section 3.5 describes the methods used to select the radionuclides of primary concern in the TSD. Section 3.6 presents the screening methods used to select the specific exposure scenarios, pathways, and biological endpoints addressed in this report. Finally, Section 3.7 summarizes the results of the screening process and the limitations of the TSD due to the constraints placed on the scope of the analyses provided in this report.

3.1 PRIMARY PURPOSE OF THE TSD

The TSD provides technical background information for the Agency to use in its evaluation of the potential for recycling scrap metal from nuclear facilities and the need for regulatory action. The information contained in this document also supports the cost-benefit analyses conducted by EPA as part of this evaluation. The TSD may also serve as the technical basis for a Regulatory Impact Analysis (RIA) done by EPA to support a future rulemaking concerning the recycling of scrap metal from nuclear facilities.

The TSD includes the following background information:

1. *Characterization of the potential sources of scrap metal that may be affected by a future EPA rule addressing the free release of scrap metal from nuclear facilities.*

   The management and disposition of scrap metal that has been generated to date, and that will be generated in the future by nuclear facilities in the U.S., is currently governed primarily by DOE and NRC regulations and guidelines that do not apply specifically to the free release of scrap metal from these facilities. As such, decision making regarding the disposition of scrap metal has not been based on a single set of comprehensive national standards. Should EPA establish such standards, behavior regarding the disposition of scrap metal from nuclear facilities may change, resulting in certain costs and benefits that need to be characterized. As a first step in gaining insight into these potential costs and benefits (which are addressed in detail in IEC 97), an understanding is needed of the sources, quantities, types, and radiological characteristics of the scrap metal that may be impacted by national standards governing the free release of scrap metal from nuclear facilities.

2. *Estimates of the potential normalized annual dose and normalized lifetime risk to the reasonably maximally exposed individual (RMEI) associated with the free release of scrap metal from nuclear facilities.*

   The term "normalized annual dose" to the RMEI refers to the high end annual effective dose equivalent (EDE) that may be received by an individual due to the release of scrap metal that may contain trace levels of radioactivity. The normalized dose is expressed in units of mrem/y EDE per picoCurie per gram (pCi/g) of a specific radionuclide in scrap metal.

   The normalized dose is a useful metric because for any free release criterion established in units of mrem/y, the normalized dose can be used to derive the average[1] radionuclide concentration level in scrap metal that corresponds to the criterion. For example, Table 7-1 indicates that the normalized dose to the RMEI from Co-60 is 0.899 mrem/y per pCi/g. Assuming a free release criterion of 1 mrem/y, the average Co-60 contamination level in scrap metal that corresponds to 1 mrem/y can be estimated by dividing the free release criterion (1 mrem/yr) by the normalized dose (0.899 mrem/yr) or 1.1 pCi/g. Alternatively, if the average concentration of a given radionuclide in scrap metal is known, the annual dose to the RMEI resulting from its release can be estimated by multiplying the normalized dose by the average concentration of the radionuclide in the scrap metal.

The normalized risk is similar to the normalized dose except that it is expressed in units of lifetime risk of cancer per year of exposure per pCi/g of a given radionuclide in scrap metal. As such, it can be used to derive that concentration of a given radionuclide in scrap metal that corresponds to a given risk. It can also be used to derive the potential lifetime risk of cancer for the RMEI associated with the release of scrap metal containing a known average concentration of a given radionuclide.

Imbedded in both these definitions is the concept of the RMEI. As used in this report, the RMEI refers to that individual, within the group of people that have the greatest potential for exposure to residual radioactivity contained in scrap metal released from nuclear facilities, who would receive the high end exposure. This group of people, which can be referred to as the critical or limiting population group for a given radionuclide, have job responsibilities or living habits that result in elevated potential for exposures as compared to other groups. Within the group, there is variability among the members with regard to their individual potential for exposure. The RMEI is that individual within the group that has a relatively high potential (e.g., 90th percentile) for exposure. As such, it is unlikely that many individuals within or outside the group could receive exposures significantly greater than those of the RMEI; most individuals that may be exposed are likely to receive exposures that are substantially lower than those received by the RMEI.

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[1] When deriving the normalized annual dose to the RMEI, the volume of scrap metal over which the radionuclide concentrations are averaged differs depending on the exposed individual. For example, when deriving the normalized annual dose to a mill worker, the volume of scrap metal of concern is the entire scrap metal throughput at the mill over a year. However, when deriving the normalized annual dose to a user of a product made from scrap metal from a nuclear facility, the averaging volume is the volume of scrap metal required to make the product.

3. *Estimates of the potential normalized, time-integrated collective dose and normalized, time-integrated potential collective public health impacts associated with the release of scrap metal from nuclear facilities.*

   The normalized, time-integrated collective dose is expressed in units of the collective dose (i.e., person rem) to which a population is committed per unit activity contained in released scrap metal. The units can therefore be simply expressed in terms of person rem per Curie, for example. The normalized collective dose is a convenient metric because, once a determination is made of the total quantity and radionuclide inventory that is to be released, the collective dose can be determined by simply multiplying the curie inventory associated with the scrap metal by the normalized collective dose for each radionuclide.

   The time-integrated normalized collective risk is similar to the time-integrated normalized collective dose except that it is expressed in units of the potential numbers of adverse health effects per Curie of each radionuclide contained in the released scrap metal. As such, it can be used to derive the potential number of health effects that may be attributable to the release of a given quantity of scrap metal containing a known inventory of radionuclides.

   As is the case for the normalized dose for the RMEI, the time-integrated normalized collective dose for each radionuclide is derived to ensure that the potential collective health impacts are not underestimated; i.e., each represents an upper end estimates given their uncertainties.

4. *Estimates of the minimum detectable concentrations of radionuclides contained within or on the surface of scrap metal.*

   The minimum detectable concentrations (MDC) of various radionuclides are needed to assess the feasibility and implementation costs of alternative free release criteria. For example, Table 8-6 reveals that the MDC for Co-60 using conventional radiation survey techniques is about 470 dpm/100 cm^2 for large area sources. This value corresponds to about 0.5 mrem/y.

3.2 PRIMARY SCREENING CRITERIA

The following screening criteria were used to focus the scope of the EPA's investigations:

1. *Sources and types of scrap metal and exposure scenarios, pathways, and biological endpoints that are potentially significant or limiting in terms of individual and collective doses and risks.*

For the purposes of its evaluation, EPA defined significance in terms of: the quantity of metal, the likelihood that the scrap metal is contaminated, the half-life of the contaminants, the concentration of radionuclides in or on the scrap, the magnitude of the potential for exposure based on a person's occupation and living habits, and the likelihood of a given adverse health impact occurring.

2. *The potential market value of the metal.*

The potential market value of a metal may influence the economic impact of a future EPA rule governing the release of scrap metal from nuclear facilities. The potential market value is determined by the product of the quantity of the scrap metal and the unit quantity price of the metal.

3.3 SOURCES OF SCRAP METAL CONSIDERED - ADMINISTRATIVE AND FUNCTIONAL CATEGORIES

The potential sources of scrap metal can be categorized by function (e.g., reactors, research laboratories) and by the administrative authority responsible for their management and disposition (e.g., DOE, NRC). Information pertaining to scrap metal from nuclear facilities is defined and accessible in terms of these two broad categories. The process used to screen potential sources of scrap metal was to: (1) review data available within each category of administrative authority and (2) assess the degree to which the functional categories were represented. This approach was found to be the most practical method for acquiring scrap metal data because the needed information was more readily accessible by administrative authority.

3.3.1 Administrative Authorities

The principal administrative authorities responsible for the management of scrap metal from nuclear facilities are:

1. The Department of Energy
2. The Nuclear Regulatory Commission
3. The Department of Defense
4. State or Superfund Authority

Table 3-1 presents an overview of the various administrative categories of sites containing or contaminated with radioactive materials. A review of data available characterizing contaminated structures within these administrative categories is provided in EPA 96 (page E4-7). The review revealed that there are a total of about 30,000 structures in the major jurisdictional sectors, of which about 8,000 are contaminated. These structures, along with scrap metal already in storage at many DOE sites, represent the major potential sources of scrap metal at nuclear facilities in the U.S.

The Department of Energy (DOE)

DOE is responsible for cleaning up more than 130 contaminated facilities in over 30 States and territories (DOE 95, BEMR Executive Summary, page iii). These include approximately 45 national laboratories and nuclear weapons production and testing facilities where environmental restoration and waste management activities are taking place. Many of these are large sites with facilities that have been used for multiple activities related to nuclear weapons research, production, and testing over the years and have many areas of contamination. Many of these facilities also have extensive mixed waste contamination. Several DOE facilities have literally hundreds of areas that are being investigated and cleaned up separately. For example, DOE's Hanford facility, which encompasses 570 square miles, is divided into about 1,100 individual "waste site units" based on their waste characteristics or other factors. EPA 96 estimates that DOE sites contain a total of 3,179 containment structures and 1,179 buildings. These sites, which have been grouped into 78 operable structures, are believed to represent the major sources of scrap metal under the authority of the DOE.

DOE's Environmental Restoration and Waste Management (EM) program is in the process of characterizing and decontaminating and decommissioning these facilities and restoring the environmental conditions at these sites. Information on the status of these programs is provided in many DOE core documents (DOE95; 95a; 95b). In addition, DOE's "Inventory Report" has estimated the current and projected inventory of potential scrap metal at many of its facilities (MIN 96). Table 3-2 summarizes these estimates.

Table 3-1. Inventory of Sites That are Known to be Contaminated with Radioactivity (from EPA94).

AGENCY/SITE TYPE	NUMBER
Federal	
DOE	
Major DOE Facilities	12[1]
National Laboratories	7[2]
FUSRAP*	27
UMTRAP**	10
Other DOE Sites	31
DOD	
Major DOD Facilities	1[3]
Sites with Burial Areas	85
Sites with Accident Contamination	1
Sites with DU*** Contamination	15
Other DoD Sites	57
Other Federal Sites	2[4]
NRC/Agreement State Licensees	
Nuclear Power Plants	125
Test and Research Reactors	63
Other Fuel Cycle Facilities	65
Rare Earth Extraction Facilities	22
Byproduct Material Facilities	4401
Non-Federal NPL Sites	
Municipal Landfills	3
Radium Sites	7
Other Sites	11
Other State Sites	- (No reliable data)

[1] Fernald, Hanford (4 subsites, including 100, 200, 300, and 1100 Areas), INEL, Mound, Nevada Test Site, Oak Ridge, Paducah, Pantex, Portsmouth, Rocky Flats, Savannah River, Weldon Spring.

[2] Argonne, Brookhaven, Fermi, Lawrence Berkeley, Lawrence Livermore (main Area and 300 AREA), Los Alamos, Sandia.

[3] Aberdeen Proving Ground

[4] Watertown Arsenal (GSA), Fremont National Forest (USDA)

* Formerly Utilized Sites Remedial Action Program

** Uranium Mill Tailings Remedial Action Program

*** Depleted Uranium

Table 3-2. Estimates of Existing and Projected Potential Sources of DOE Scrap Metal Used for Screening Purposes (from MIN 96)

DOE Site	Tons of Scrap Metal		
	Current Inventory	Projected from D&D	Total
Fernald	23,100	149,000	172,000
Hanford	16,263	129,000	145,000
INEL	2317	37,000	39,300
K-25	45,000	-	45,000
ORO	1764	727,000	729,000
Paducah	32,400	-	32,400
Portsmouth	24,600	-	24,600
LANL	3407	-	3407
Rocky Flats	70	29,000	29,070
SRS	16,461	4000	20,461
Pinellas	54	-	54
Weldon Spring	30,623	-	30,623
Y-12	11,338	-	11,338
Total	207,370	1,073,335	1,200,000

Based on this understanding of the potential quantities of DOE scrap metal, the DOE sites and facilities listed in Table 3-2, with the exception of Pinellas (due to the very small quantity), are explicitly included in the scope of the TSD[2].

The Nuclear Regulatory Commission (NRC)

The NRC and its Agreement States have licensed about 22,000 facilities for the production and handling of radioactive materials (EPA 93). About one third of these are NRC licensees, while

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[2] Other sections of this report present more detailed characterizations of the quantities of scrap metal at these sites. The results of these more detailed investigations provide volumes that differ from those in Table 3-2. However, it was the quantities of potential scrap metal summarized in Table 3-2 that were used to identify those sources of DOE scrap metal that would be included in this TSD. As indicated in other sections of the TSD, estimates of scrap metal quantities are continually being revised.

the remainder are licensed by Agreement States under Section 274 of the Atomic Energy Act. Licensees include universities, medical institutions, radioactive source manufacturers, and companies that use radioisotopes for industrial purposes. About 50 percent of NRC's 7,500 licensees use either sealed radioactive sources or only small amounts of short-lived radioactive materials. Activities at these facilities are not likely to result in residual radioactive contamination that will need to be cleaned up and disposed because: (1) the radionuclides remain encased and cause little (if any) contamination and/or (2) because the radionuclides rapidly decay to non-radioactive elements. A small number of licensees (e.g., radioactive source manufacturers, radiopharmaceutical producers, and radioactive ore processors) conduct operations that could result in substantial radioactive contamination in portions of the facility. In addition, about 250 facilities associated with the production of nuclear power[3] maintain large inventories of radioactive materials; many of these facilities will need to be cleaned up before their licenses can be terminated.

EPA 96 estimates that NRC licensees are responsible for a total of 4,625 buildings and structures, including:

- 125 nuclear power plants
- 63 test reactors
- 930 sealed source manufacturers
- 3,471 medical and R&D facilities
- 22 rare earth production facilities
- 14 fuel fabrication plants

Only the scrap metal from commercial nuclear power plants are explicitly included in EPA's current assessment. The other potential sources are not addressed due to the relatively small volume of scrap metal generated and/or the short half-lives of the radionuclides involved.

Department of Defense (DoD)

DoD's Installation Restoration Program (IRP) consists of over 17,500 potential hazardous waste sites located at 1,877 installations (Baca 92). DoD sites vary widely in function and size. They include hospitals, laboratories, proving grounds, bombing and gunnery practice ranges, missile

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[3] These include nuclear power plants, non-power (research and test) reactors, fuel fabrication plants, uranium hexafluoride production plants, uranium mill facilities, and independent spent fuel storage installations.

launch sites, weapons manufacturing and storage facilities, and reactors. Only a few of these are currently known to have radioactive contamination although these sites have not been fully characterized. Consequently, it is not possible to reliably estimate the number of sites with radioactive contamination.

DoD sites may contain small enclosed radiation sources, such as radium and tritium instruments. They may also contain larger sources, such as research reactors, and dispersed sources, such as laboratory waste storage areas and test ranges contaminated with plutonium and fission products.

EPA 96 estimates that DoD is responsible for 161 contaminated structures, including 43 containment structures and 118 buildings. Due to the relatively limited potential for scrap metal and the unavailability of data characterizing the scrap metal, DoD sources of potential scrap metal are not explicitly addressed in the TSD.

State or Superfund Authority

This administrative category includes sites that are not licensed by NRC or Agreement States but are under State or Superfund authority. (Sites that are under Superfund authority are those that are on the National Priority List and are being cleaned up by the Federal government.) This category includes about 1,000 particle accelerator sites that generally contain only small amounts of short-lived residual radioactivity after shutdown. Other sites included in this category contain long-lived naturally-occurring radionuclides varying from small packaged radiation sources to large areas of mostly low-level dispersed contamination, including mining wastes and materials, tailings from ore processing, and residues from university or commercial research activities.

The principal sources of scrap metal in this administrative category include metal contaminated with naturally-occurring radioactive material (NORM), primarily from the oil and gas industry. This source of scrap metal has been excluded from EPA's current assessment because the Agency has chosen to limit its evaluation to scrap metal generated by nuclear facilities, thereby excluding sources contaminated with NORM.

3.3.2 Functional Categories

It is also possible to categorize scrap metal from nuclear facilities according to a variety of functional categories. Table 3-3 presents the functional categories used to classify nuclear facilities in EPA 94. This categorization scheme is useful because it helps to assess the degree to which the compilation of data characterizing scrap metal represents a full range of functional categories.

Table 3-3. Functional Categories for Nuclear Facilities and Sites Containing or Contaminated with Radioactive Materials

Functional Category	Administrative Authority	Represented in the Volume of Scrap Metal Addressed in this Report
Mines, Mills, and Rare Earths	DOE and NRC Licenses	No**
Conversion and Enrichment	DOE	Yes*
Fuel Fabrication and Weapons Assembly	DOE, DoD, and NRC Licensees	Yes*
Reprocessing and Extraction	DOE and NRC Licensees	Yes*
Reactors	DOE and NRC Licensees	Yes
Research, Biomedical, and Analytical Labs	NRC Licensees	No
Industrial and Commercial (Non-sealed sources)	NRC Licensees	No
Sealed Source Users	NRC Licensees	No
Nuclear Medicine Departments	NRC Licensees	No
Accelerators	DOE	No
Fusion Facilities	DOE	No
Nuclear Test Sites	DOE	No
Weapons Accidents and Safety Tests	DOE and DoD	No
DU	DoD	No
Other DoD Facilities	DoD	No
Waste Disposal	DOE, DoD, NRC Licensees, and EPA Non-Federal NPL Sites	No
Naturally Occurring Radioactive Material (NORM) Wastes Mixed with AEA Materials	DOE and EPA Non-Federal NPL Sites	No**
Entire Multipurpose Facility	DOE and DoD	Yes*

* To the extent captured by Table 3-2

** Not within the scope of EPA's current investigations.

3.4 TYPES OF SCRAP METAL CONSIDERED

Not all scrap metal within the administrative and functional categories evaluated by EPA was considered in this assessment. As mentioned in Chapter 1, the Agency's evaluation focussed on scrap metal that is moderately contaminated with radioactivity as a result of deposition or neutron activation. Some metal is so highly contaminated that recycling is not feasible. For example, the canyons at fuel reprocessing facilities and reactor internals are so contaminated that they are not considered potential sources of recyclable scrap metal and were therefore not included in this assessment. On the other hand, scrap metal that is not radiologically contaminated, such as scrap from locations outside the radiation control areas at DOE sites and commercial nuclear power facilities, is also not included.

The types of metals that EPA explicitly considered have either economic significance and/or potential public health significance. Economic significance is determined simply by assessing the potential quantity and unit quantity value of the metal. The potential public health significance is a more complex problem. Public health significance is determined by evaluating the potential for the metal to result in significant radiation exposures to either a selected group of individuals or the general population. This means that, though the quantity or value of a potential type of scrap metal may be small, the radionuclide content and the way in which the metal is processed, handled, and ultimately used commercially after being released from a nuclear facility could result in the limiting normalized doses for specific radionuclides. As such, such metals require explicit consideration when developing the normalized dose and risk tables.

3.4.1 Screening Based on Economic Value

Table 3-4, taken from data provided in MIN 96, presents an overview of the potential economic value of metals that are known to be potential sources of scrap metal from nuclear facilities. Based on potential value and consideration of the availability of data characterizing each metal type, the metals evaluated include:

- Carbon steel
- Stainless steel
- Copper
- Aluminum
- Nickel

Data characterizing the potential quantities and radiological characteristics of the selected metals were gathered (where available) for the facilities and sites identified in Section 3.3. However, as discussed in subsequent sections, normalized doses and risks were developed only for carbon steel.

Table 3-4. Potential Economic Value of Types of Scrap Metal (MIN 96)

Metal	Volume (tons)			Unit Value ($/ton)	Total Value ($)
	DOE	NRC	Total		
Carbon Steel	0.7E6	3.5E6	4.2E6	80	336E6
Stainless Steel	0.4E5	1.9E5	2.3E5	600	138E6
Gal. Iron	-	1.4E5	1.4E5	80	11.2E6
Copper	6.1E4	7.7E4	1.38E5	2,400	331E6
Inconel	-	1.3E4	1.3E4	about 1000	130E6
Lead	2.0E3	5.2E3	7.2E3	250	1.8E6
Bronze	-	2.8E3	2.8E3	-	-
Aluminum	2.9E4	0.2E4	3.1E4	450	14E6
Brass	-	1.1E3	1.1E3	-	-
Nickel	5.9E4	1E2	5.91E4	4950	293E6
Silver	-	1E2	1E2	256,000	25.6E6
Depleted Uranium	1.9E5 Ft3	-		-	-
Others	3.1E5	-		-	-
Total	1.2E6	3.9E6	5.2E6		about 1.3 billion

3.4.2 Screening Based on Public Health Considerations

Notwithstanding the value of a given type of metal, a metal may also be of interest if it has the potential to have normalized doses and risks which are substantively greater than those for carbon steel. The mathematical models and assumptions used for deriving normalized doses and risks, as described in Chapter 6, are based on a reference electric arc furnace used to recycle carbon steel. The normalized doses for the Reasonably Maximally Exposed Individual (RMEI) presented in Table 7-1 are based on assumptions regarding the throughput and dilution of scrap metal representative of recycling carbon steel. In addition, the assumptions regarding partition factors, geometries, exposure scenarios and end uses of the slag, baghouse dust and commercial products resulting from recycling scrap metal are based on carbon steel. It is possible that recycling aluminum, copper, or other metals from nuclear facilities could result in normalized doses and risks that are substantially higher than those derived for carbon steel. In other words, it is possible that the free release of metals other than carbon steel, based on normalized RME doses developed for carbon steel, could result in doses to the RMEI which might exceed any future radiation protection standards. Similarly, it is possible that the normalized collective doses for other metals are greater than those derived for carbon steel. These issues are discussed further in the following subsections.

Normalized RME Doses

In order to address the applicability of normalized RMEI doses and risks for carbon steel to other metals, a review was performed of the processes used to recycle copper and aluminum. Appendices B and C present the results of this review. The review attempted to ascertain whether there are aspects to the recycling of aluminum and copper that are significantly different than those associated with the recycling of carbon steel and which could result in substantively higher normalized RMEI doses for some radionuclides. It was determined that the processes are very different and the potential exists for markedly different dilution and partition factors for some metals which could influence the normalized RMEI doses. Hence, specific investigations are planned to further assess this issue.

Normalized Collective Doses

The collective doses are derived as the product of the normalized collective dose (person rem/Ci) for a given radionuclide and the total quantity of that radionuclide that is contained in the scrap metal. Table 3-4 reveals that the quantity of metal other than carbon steel that may be released from nuclear facilities is a very small fraction of the quantity of carbon steel likely to be released. This is especially true for non-ferrous metals. Consequently, the potential for metals other than carbon steel to contribute significantly to the collective dose, relative to that of carbon steel, is extremely small. It is, therefore, not necessary to derive normalized collective doses for metals other than carbon steel.

3.5 RADIONUCLIDES SELECTED FOR CONSIDERATION

The criteria used to select the radionuclides for which normalized doses and risks were derived are as follows:

1. Half-lives greater than 6 months. This cutoff was selected because the time delay between contamination and release from the facility would preclude significant exposures associated with the shorter lived radionuclides. Most radionuclides with half-lives less than 6 months, and which appear to have the potential to be present in scrap metal, have very short half-lives. Hence, 6 months was selected as a convenient break point.
2. Presence in scrap metal. The relevant radionuclides were identified by a review of source documents describing the radionuclides associated with the uranium fuel cycle and found in scrap metal.

A detailed discussion of this selection process is presented in Appendix D.

3.6 SCENARIOS, PATHWAYS, AND BIOLOGICAL ENDPOINTS CONSIDERED

There are about 160 mills in the United States that may be willing to accept scrap metal from nuclear facilities for recycling if radiation protection standards were established. Theoretically, in any given year, some of these mills might handle large quantities of scrap metal from nuclear facilities, while others might handle little to none. The number of people that may be exposed to radioactivity as a result of these activities could be numerous. They include the handlers and transporters of the scrap metal, the various workers at the mills, other workers downstream from the mill operations who handle the steel products, slag, and other materials contaminated as a result of recycling scrap metal, members of the public who live near the mills and are exposed to the effluents from the mills, consumers who use the products made from scrap metal, and people exposed as a result of the final disposal of the products and wastes associated with the recycling of scrap metal.

The workers at the mills are likely to receive the highest doses. Some workers will receive the highest doses because of the nature of their jobs and, perhaps, some unique aspects of the operations at specific mills. The individuals that are anticipated to receive the highest individual doses as a result of one year of recycling operations are referred to as the RMEI. Hence, out of a population that may be exposed as a result of one year of scrap metal recycling operations, the RMEI is anticipated to receive the highest dose.

A screening process was used to identify those categories of people that have the greatest potential for individual exposure, the pathways responsible for the exposure, the key assumptions used to model the exposures, and the biological endpoints of concern. The selection of the scenarios, exposure pathways, and biological endpoints addressed in this report were selected based on a review of previous reports addressing the recycle of scrap metal (see Appendix D), several visits to scrap yards and steel mills, and the insight gained in the preparation of the precursor documents to this report, primarily SCA 95.

Screening analyses performed in support of SCA 95 derived normalized doses for about 70 individuals that were grouped according to 14 categories:

1. Salvage yard operations
2. Transportation to mill
3. Scrap yard operations at mill
4. Furnace operations
5. Foundry operations
6. Bag house operations
7. Slag operations at mill
8. Transporting slag
9 Transporting baghouse dust
10. Slag use in road construction
11. Manufacturing products made from recycled scrap metal
12. Distribution and use of products made from recycled scrap metal
13. Disposal of slag, baghouse, dust and product
14. Offsite population exposed to airborne emission

The pathways of exposure included direct radiation, inhalation, ingestion of soot, ingestion of farm products contaminated as a result of airborne emission from the mill, and contamination of ground water due to leachate from slag piles.

As a result of these screening analyses, the 17 persons and scenarios described in Section 5 (Table 5-1) were selected for explicit consideration in this report. In addition, offsite doses from the food ingestion pathways for C-14 and I-129 were added to the list since these radionuclides have the potential to be released in the gaseous effluent from a mill and reconcentrate in vegetable, beef, and milk produce.

The values selected for the modeling parameters used in the analyses (see Chapters 5 and 6) were selected to represent the upper end of the range of possible values.[4] Among the number of mills and numerous scrap dealers, there is large variability in working practices, exposure durations, airborne dust loadings, etc. As such, by selecting upper end assumptions, there is a level of assurance that the normalized doses thereby derived reflect the worker and operational practices that tend to result in the higher normalized doses. However, in selecting the modeling assumptions, the extreme upper ends of the possible range of the distributions were not selected in order to avoid over-conservatism. Hence, there may be some facilities, under some circumstances, and some time periods, that could have normalized doses somewhat higher than the derived values.

The biological endpoints of potential concern in the recycling of scrap metal from nuclear facilities could include carcinogenic, genetic, and teratogenic effects. The methodology used by EPA in its evaluation was designed to specifically quantify the risk of total cancers (not including non-fatal skin cancer) and fatal cancers. This methodology is consistent with the approach typically taken by EPA in developing its radiation protection standards. The Agency does not quantify the potential for non-carcinogenic health effects because they are far less likely to occur than carcinogenic effects at the dose levels potentially associated with recycling scrap metal from nuclear facilities (EPA 89). This approach is supported by current international radiation protection guidance (UN 93).[5]

Though the models were developed to explicitly quantify the potential number of carcinogenic effects, the models also present the time integrated collective dose (person rem). This parameter can be used to estimate potential stochastic effects, other than carcinogenesis, that may be averted by future radiation protection standards governing the recycling of scrap metal from nuclear facilities.

The objective of the screening process was to limit the individuals, scenarios, pathways, and biological endpoints to a manageable number without excluding any individuals, scenarios,

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[4] It is not possible to assign quantitative confidence limits at this time based on the limited data that is currently available.

[5] UN 93 cites a risk coefficient of 5×10^{-4} per rem for lifetime fatal cancer risk in a nominal population of all ages. The risk coefficient cited for genetic effects is 1.2×10^{-4} for a reproductive population for all generations after exposure. For clinically important disorders for the first generation of offspring of exposed parents, the genetic risk coefficient is cited as 0.2×10^{-4} to 0.4×10^{-4} per rem for the reproductive part of the population.

pathways, and biological endpoints that could result in substantially higher normalized doses than those presented in Table 7-1. In addition, the modeling assumptions were selected to provide a level of assurance that the actual normalized doses at real facilities are not significantly underestimated.

Multiple Pathways

Consideration was given to the assumption that some individuals could be exposed by multiple pathways/scenarios. For example, the lathe operator may be exposed to the lathe at work and to kitchen appliances at home or he may live downwind from a mill recycling scrap metal from nuclear facilities. Such combinations of limiting pathways and scenarios are unrealistic. In the first example, not only would the lathe be made entirely from scrap from nuclear facilities, but so would the lathe operator's kitchen appliances. The probabilities of this occurring are extremely small. It must be recognized that, in reality, during the approximate 10 to 50 year time period that the approximate 2 to 3 million metric tonnes of scrap metal from nuclear facilities may be recycled, the metal will be diluted in a national scrap metal flow of about 68 million tons (62 million metric tonnes) per year. Hence, the scrap from nuclear facilities would experience an approximate 200-fold dilution in the finished product. The assumption that the lathe operator is exposed to a lathe made entirely from undiluted scrap metal from the nuclear industry is itself extreme, and the probability that he is also exposed to other undiluted products is even less likely.

The second example given is the lathe operator who, by chance, happens to live downwind from the mill that receives the most scrap metal from nuclear facilities in the country. The limiting mill was selected as a mill that receives 13 percent of its scrap in one year from nuclear facilities. Appendix G indicates that there may be one mill where this can occur, but it is much more likely that most mills, if not all, will receive a much smaller percent. Hence, the combination of assumptions that the lathe operator not only has a lathe that is made entirely from scrap from nuclear facilities, together with the assumption that he happens to live downwind from the limiting mill, is unrealistic.

Similar arguments can be made regarding other limiting scenarios and pathways. Because of the unlikelihood that they will occur, combined pathways/scenarios are not assumed in this analysis.

Other Pathways

Many scoping analyses were performed to ensure that important scenarios and pathways were not overlooked in EPA's evaluation. For example, consideration was given to including exposure from food grown in soil that uses slag as a soil conditioner (liming agent) in determining the RMEI normalized dose. Scoping calculations revealed that, though this is a realistic pathway, it could not be limiting for the RMEI because of the large dilution that would be experienced by the slag when used as a soil conditioner. For example, assuming 100 pounds of slag are applied per 1000 ft^2, down to a plow depth of 15 cm, the combined vegetable/meat/milk pathway normalized dose for Sr-90 in slag would be less than 0.1 mrem/yr per pCi/g of Sr-90 in scrap metal. This is as compared to the limiting normalized dose for Sr-90 of 3.03 reported in Table 7-1. However, this scenario is significant with regard to the collective impacts and is included in that analysis.

3.7 SUMMARY OF THE SCREENING PROCESS AND ASSOCIATED LIMITATIONS

This section summarizes the results of the screening process and the resulting scope of the analyses that comprise EPA's evaluation. This section also includes a brief discussion of the limitations associated with the selected scope. "Limitations" refers to potential misleading or erroneous conclusions that could result from limiting the scope of the analyses.

3.7.1 Sources of Scrap Metal

Scope

Out of the approximate 17 functional categories (see Table 3-3) and the four major administrative categories that represent the sources of scrap metal from nuclear facilities, the EPA's analysis explicitly addresses four functional categories, two administrative categories, and 14 sites that contain or are contaminated with radioactivity. The four functional categories include DOE conversion and enrichment facilities, fuel fabrication and weapons assembly plants, reprocessing and extraction facilities, and reactors. The two administrative categories are DOE and the NRC. The 14 sites include the 12 DOE sites listed in Table 3-3 (Pinellas is not included), one reference BWR (Washington Public Power Supply System Nuclear Project No. 2) and one reference PWR (Trojan Nuclear Plant).

Limitations

The Agency believes that the important sources in terms of quantity of scrap metal available for recycle have been captured in the analysis. Hence, a cost/benefit analysis based on the sources addressed in this report should provide a reasonable national perspective on the potential impacts of recycling of scrap metal from nuclear facilities. However, the analysis will provide only limited insight into the variability of the cost/benefits associated with individual sites and categories of facilities. As such, unique site-specific issues may need to be addressed on a case-by-case basis.

> *Limitation 1 - The analyses will provide only limited insight into the variability of the cost/benefits associated with individual sites and categories of facilities.*

3.7.2 Types of Scrap Metal from Nuclear Facilities

Scope

Of the approximate 12 types of metals and metal alloys that may comprise scrap metal from nuclear facilities, the Agency has characterized the volume and radionuclide composition of five: carbon steel, stainless steel, copper, aluminum, and nickel. Of these, normalized doses are only derived for carbon steel, which may be considered generally representative of ferrous metals.

Limitations

The five types of metal selected for characterization represent the majority of the potential quantity of scrap metal available from nuclear facilities for recycle. Hence, there is very little likelihood that an important type of scrap metal has been overlooked from the perspective of quantity or economic value. Therefore, the analysis should provide a reliable national perspective on the costs and benefits of recycling different types of metals from nuclear facilities. However, the analyses provide limited information pertinent to the assessment of the impacts of recycling metals not explicitly addressed in this report. As a result, conclusions regarding the cost/benefit of recycling metals presented in this report cannot necessarily be extended to metals outside the scope of EPA's analyses. Hence, the applicability of the results of the Agency's evaluation to other metals may need to be addressed on a case-by-case basis.

> *Limitation 2 - Conclusions regarding the cost/benefit of recycling the metals presented in this report cannot necessarily be extended to metals outside the scope of EPA's analyses.*

The normalized doses presented in the TSD apply specifically to carbon steel. They cannot necessarily be extended to other metals, nor is there assurance that the doses from other metals contaminated with specific radionuclides may not be more limiting than those from carbon steel.

> *Limitation 3 - The normalized doses and risks presented in the TSD for carbon steel cannot necessarily be extended to other metals, nor is there assurance that normalized doses from other metals contaminated with specific radionuclides may not be more limiting than those for carbon steel.*

3.7.3 Radionuclides

Scope

127 radioisotopes could, in theory, be present in scrap metal from nuclear facilities. Of these, the TSD explicitly addresses 40.

Limitations

A formal process was used to screen to the possible radionuclides of concern and to select the 40 that were ultimately evaluated. Hence, the possibility that one or more significant radionuclides has been overlooked is small. However, the 6 month half-life cutoff was based on an engineering judgment that there will be a substantial delay (on the order of years) between the contamination of scrap metal at nuclear facilities and its release as scrap metal. If the time between contamination and free release is substantially less than a few years (e.g., months), the list of radionuclides addressed in the analysis may need to be expanded to assess these special cases.

> *Limitation 4 - The list of radionuclides addressed in the analysis may need to be expanded for sites where the time between contamination and release of scrap metal is on the order of months instead of years.*

3.7.4 Scenarios, Pathways, Modeling Assumptions, and Biological Endpoints

Scope

Out of the virtually unlimited number of possible exposure scenarios, 19 scenarios were selected for analysis. The pathways selected for analysis include external exposure, inhalation of dust, ingestion of soot, and ingestion of produce. Upper end values for the modeling assumptions were selected. Of the range of biological endpoints that could be of concern (i.e., individual and collective dose, individual risk of fatal and non-fatal cancer, total numbers of fatal and non-fatal cancers in a population, hereditary effects in individuals and populations, and teratogenic effects in individuals and populations), hereditary and teratogenic effects were not explicitly addressed.

Limitations

There is always a possibility that some individuals, at some facilities, could have working practices that could result in higher normalized doses than those identified for the RMEI. There could also be some pathways, other than those addressed in this report, that could also result in higher normalized doses. In addition, though a deliberate effort was made to select high end modeling assumptions, there could be some facilities where the exposure durations are longer and the dust loadings factors are higher than those considered in the analysis. These limitations are not unique to this analysis; therefore, they are not identified as limitations specific to EPA's current evaluation of the impacts of recycling scrap metal from nuclear facilities.

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CHAPTER 4

QUANTITIES AND CHARACTERISTICS OF POTENTIAL SOURCES OF SCRAP METALFROM DOE FACILITIES AND COMMERCIAL NUCLEAR POWER PLANTS

This chapter provides quantitative data on the amount of scrap metal potentially available for recycle from Department of Energy (DOE) sites and the commercial nuclear power industry. Scrap metal quantities for DOE sources and the means by which the data were developed are discussed in Section 4.1 of this chapter. Because data available for the nuclear power industry are considerably more detailed, only a summary description of the scrap metal database for power reactors is presented in Section 4.2 of this chapter. A comprehensive discussion of scrap metal sources that will be generated from the decommissioning of commercial reactors is provided in Appendix A of the TSD.

4.1 EXISTING AND FUTURE SCRAP METAL QUANTITIES AVAILABLE FROM DOE

4.1.1 Background Information

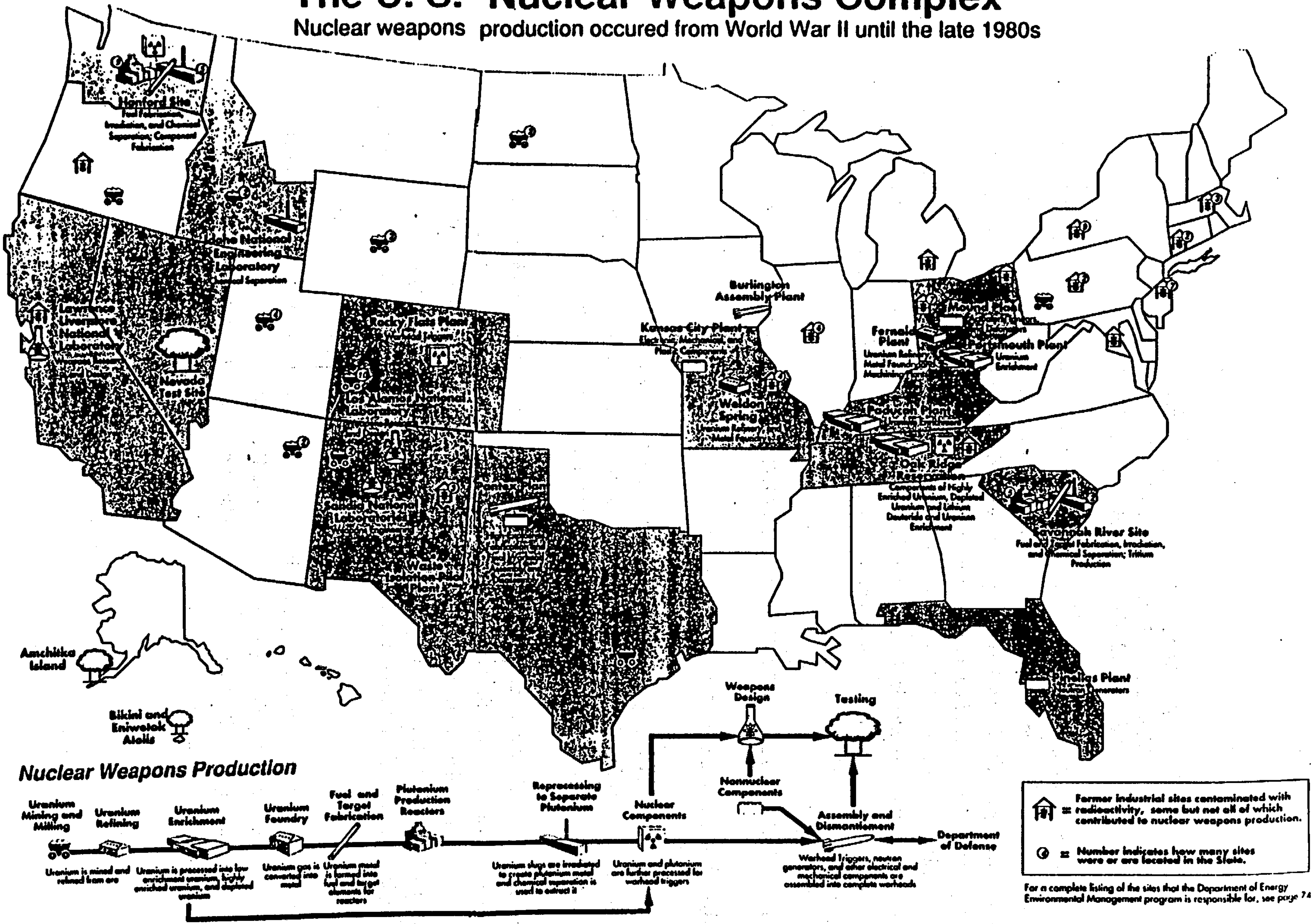
The DOE designs, tests, manufactures, and maintains nuclear weapons that are central to the national security of the United States. This effort started with the Manhattan Project and the development of the first nuclear weapons that were employed in World War II.

Shortly after World War II, deteriorating relations between the United States and the Soviet Union led to a massive nuclear arms race. In the United States, the nuclear arms race resulted in the development of a vast research, production, and testing network of Federal facilities that came to be known as the "nuclear weapons complex." During half a century of operations, the complex manufactured tens of thousands of nuclear warheads and test-detonated more than one thousand.

During its peak, this complex consisted of 16 major facilities each with its own specific mission (Figure 4-1). Weapons production stopped in the late 1980's initially to correct environmental and safety problems. With the end of the Cold War, most of the nuclear weapons activity has been suspended indefinitely.

The U. S. Nuclear Weapons Complex

Nuclear weapons production occured from World War II until the late 1980s



🏠 = Former industrial sites contaminated with radioactivity, some but not all of which contributed to nuclear weapons production.

● = Number indicates how many sites were or are located in the State.

For a complete listing of the sites that the Department of Energy Environmental Management program is responsible for, see page 74

Figure 4-1
(Source: DOE 95a)

With the end of the Cold War and a diminished threat, the DOE no longer needs many of the facilities and their components that formerly constituted the DOE nuclear weapons complex. DOE is now in the process of deciding what should be done with facility structures and materials that in many instances are radioactive or are radioactively contaminated. Among the materials that pose significant disposition problems are large quantities of metals that have become radioactively contaminated in various phases of extracting, testing, and producing nuclear weapon materials.

Radionuclides Associated with Nuclear Weapons

The principal components of nuclear weapons are highly enriched uranium and plutonium. Early nuclear weapons were designed to use either highly enriched uranium or plutonium that, when forced into a "critical mass," would sustain a nuclear chain reaction and result in a nuclear explosion. As designs for nuclear weapons improved, a new generation of "thermonuclear weapons" evolved that require both plutonium and uranium highly enriched with the isotope U-235. Thermonuclear weapons also require a third ingredient: tritium, a radioactive gas of hydrogen that boosts the explosive power of the nuclear weapon commonly referred to as the Hydrogen Bomb. The process by which these three components are produced is the source of radioactive contamination of scrap metals at DOE facilities and the subject of this chapter.

Enriched Uranium. In nature, more than 99 percent of uranium atoms have an atomic weight of 238 with less than one 1 percent having an atomic weight of 235. However, only uranium-235 is capable of undergoing nuclear fission that is useful in the chain reaction of nuclear weapons. To make highly enriched uranium-235, DOE facilities at the Oak Ridge Reservation in Tennessee used two elaborate processes to isolate U-235 from U-238: (1) electromagnetic separation in the "Calutron" (California University cyclotron) and (2) gaseous diffusion.

Separated and U-235 enriched uranium hexafluoride gas must be converted into a metal matrix before it is used in nuclear weapons production. At the Fernald uranium foundry in Ohio, the uranium gas was chemically converted into uranium metal. Enriched uranium metal was: (1) used as fissionable material in nuclear weapons and (2) it was fabricated into nuclear fuel that operated DOE "production reactors."

Between 1944 and 1988, DOE operated 14 plutonium-production reactors at the Hanford and the Savannah River Sites producing about 100 tons of plutonium. Plutonium-239 is produced by irradiating "depleted" uranium metal targets.

Both irradiated targets and the spent fuel of production reactors are the primary sources for the production of weapon-grade plutonium. Unfortunately, both sources also contain hundreds of different radioactive isotopes that must be chemically separated. Scientists developed elaborate physical structures and chemical processes to accomplish this separation in a manner that considered worker and public safety. A total of eight chemical separation plants, called "canyons," were operated for the DOE that employed the PUREX process for the separation and recovery of plutonium and uranium. In total, the eight chemical separation plants (i.e., canyons) generated more than 100 million gallons of radioactive wastes that are currently contained and stored at DOE facilities.

Sources of Data Used to Quantify and Characterize DOE Scrap

A thorough search for available reports and study data that might contain useful information regarding scrap metal inventories and a characterization of those inventories identified only a very limited quantity. This was not unexpected when viewed in context of the highly secretive/classified nature of past nuclear weapons activities, the relatively short time since the end of the Cold War, and the yet-undecided future for many DOE facilities.

For these reasons, the DOE has only in recent years begun to evaluate existing and future material inventories and their management. Some of DOE's earliest attempts to assess material inventories were based on the most cursory of data; data that were further compromised by an uncertain and continuously revised projection of future needs. Earlier reports are, therefore, of limited value and data reported therein have been revised and updated to reflect the most current information, facility status, and future needs.

Currently, the most informative reports pertaining to existing and future scrap metal inventories include the following:

(1) DOE/EM-0275, 1996, *A Report of the Materials in Inventory Initiative. Taking Stock: A Look at the Opportunities and Challenges Posed by Inventories from the Cold War Era*, U.S. Department of Energy, Office of Environmental Management. (Note: This report is commonly referred to as the "1996 MIN Report.")

(2) DOE/HWP-167, 1995, *U.S. Department of Energy Scrap Metal Inventory Report for the Office of Technology Development, Office of Environmental Management*, prepared by Hazardous Waste Remedial Actions Program for the U.S. Department of Energy. (Note: This report is commonly referred to as the "HAZWRAP Report.")

(3) EPA-1995 Contract Report, *Scrap Metal Inventories at U.S. Nuclear Facilities Potentially Suitable for Recycling*, prepared by S. Cohen & Associates, Inc. for the U.S. Environmental Protection Agency, Office of Radiation and Indoor Air.

(4) ES/ER/TM-171, 1995, *Gaseous Diffusion Facilities Decontamination and Decommissioning Estimate Report*, prepared by Lockheed Martin Energy Systems, Inc. for the U.S. Department of Energy.

Collectively, these four documents identified 13 DOE facilities as principal sources of scrap metal. A brief description of each of the thirteen sites is presented below.

- Fernald. Located on 1,050 acres in the southwest corner of Ohio, Fernald Environmental Management Project (formerly known as the Feed Materials Production Center) was constructed in the early 1950s to convert uranium ore to uranium metal targets. Uranium targets were subsequently shipped to DOE production reactors, which irradiated targets for the production of plutonium used in nuclear weapons. Over a 36-year period, this facility produced over 225 million kilograms of purified uranium. Production of uranium targets ceased in 1989. Principal radionuclide contaminants include uranium and its radioactive daughter products and technetium-99.

- Hanford. The Hanford reservation encompasses about 560 square miles within the Columbia River Basin in southeastern Washington and borders the Tri-Cities area of Richland, Pasco, and Kennewick to the south. Beginning in the early 1940s, nuclear materials were produced at Hanford. Activities once included plutonium production and separation, advanced reactor design and testing, basic scientific research, and renewable energy technologies development.

- Idaho National Engineering Laboratory. The Idaho National Engineering Laboratory (INEL) encompasses an area of approximately 890 square miles in southeastern Idaho on the edge of the Eastern Snake River Plain. INEL is a multipurpose laboratory supporting the engineering and operations efforts of DOE and other federal agencies in the areas of nuclear safety, reactor development, reactor operations and training, waste management and technology development, and energy technology/conversion programs. Over 50 nuclear reactors, most of them small test reactors, have existed at INEL. Some of these reactors and their associated support buildings have been decommissioned and demolished. Others are planned for decommissioning.

- Los Alamos National Laboratory. Los Alamos National Laboratory (LANL) occupies about 43 square miles approximately 25 miles northwest of Santa Fe, New Mexico. LANL was established in 1943 with the specific responsibility of developing the world's first nuclear weapon. The Laboratory's original mission rapidly broadened to include research programs in nuclear physics, hydrodynamics, conventional explosives, chemistry, metallurgy, radiochemistry, and relevant life sciences. In addition to research, a second important mission of the Laboratory between 1945 and 1978 was to process plutonium metal and alloys from nitrate solution feedstock provided by other DOE production facilities. Other operations included reprocessing of nuclear fuel, processing of polonium and actinium, and producing nuclear weapons components. Although the Laboratory has retained many of the original research programs dealing with national defense, its current mission has been expanded to include research in emerging technologies pertaining to biomedicine, space nuclear systems, materials sciences, computational sciences, and environmental management.

- Nevada Test Site. The Nevada Test Site (NTS) is 65 miles northwest of Las Vegas and occupies 1,350 square miles, making it the largest facility in the DOE complex. NTS has been the primary site for atmospheric and underground nuclear weapons testing by DOE, with more than 300 nuclear tests conducted above and below ground at NTS and at seven other locations outside Nevada. Since 1963, all U.S. nuclear weapons tests at NTS have been conducted underground. In addition to weapons testing, NTS is also used for low-level radioactive waste disposal.

- Oak Ridge National Laboratory. Founded in 1942, the Oak Ridge National Laboratory (ORNL) occupies about 2,900 acres within the Oak Ridge Reservation. The Laboratory's original mission was to produce and chemically isolate the first gram quantities of plutonium for use in nuclear weapons. With time, the scope of ORNL greatly expanded to include production of other isotopes, fundamental research in a variety of scientific disciplines, research pertaining to hazardous and radioactive materials, environmental studies, radioactive waste management/disposal, and a wide range of educational programs.

- Y-12 Plant. Built in 1943 as part of the Manhattan Project, the Oak Ridge Y-12 Plant occupies approximately 811 acres within the Oak Ridge Reservation. This facility consists of some 250 buildings that house about seven million square feet of laboratory, machining, and research and development areas. The initial mission of the Y-12 Plant, which began operation in November of 1943, was the separation and enrichment of U-235 from natural uranium by an electromagnetic separation process. When gaseous diffusion technology became the accepted process for uranium enrichment, the magnetic separators were taken out of commission in 1946. Since that time, the Y-12 Plant's mission has shifted to the disassembly of returned weapons components, quality evaluation for the existing stockpile of nuclear weapons, and supportive research in engineering designs associated with production and fabrication of nuclear weapons components.

- K-25 Site. The Oak Ridge K-25 Site occupies about 1,500 acres within the Oak Ridge Reservation. The K-25 Gaseous Diffusion Plant was built as part of the Manhattan Project during World War II to supply uranium hexafluoride for the production of highly enriched uranium for nuclear weapons production. Construction of the primary K-25 building started in 1943 and was fully operable by 1956. Exclusive production of highly enriched uranium for weapon production through 1964 was gradually replaced with commercial-grade, low-enrichment uranium production for the emerging nuclear power industry. Because of the declining demand for enriched uranium, the K-25 Plant was placed on standby in 1985 and was permanently shutdown in 1987.

- Paducah. Located in Kentucky, the Paducah Gaseous Diffusion Plant site occupies approximately 750 acres of federally-owned land. The Paducah Plant was constructed in the early 1950s for the purpose of enriching uranium by the gaseous diffusion process. Since 1991, the plant has only produced low-enriched uranium for use as fuel in commercial nuclear power plants.

- Portsmouth. The Portsmouth Gaseous Diffusion Plant is located on about 3,700 acres of federally-owned land in Ohio. In spite of the existing gaseous diffusion program at K-25 facility in Oak Ridge and Paducah in Kentucky, the Portsmouth facility was built to meet the demand for highly enriched uranium created by the emergence of nuclear submarine reactors and low-enriched uranium for projected commercial nuclear power reactors. Since 1991, the plant has produced only low-enriched uranium for use by commercial nuclear power plants, and, since 1993, production operations were assumed by the United States Enrichment Corporation, a government corporation formed under the Energy Policy Act of 1992.

- Rocky Flats. The Rocky Flats Environmental Technology Site (RFETS) covers 11 square miles located approximately 16 miles northwest of Denver. Its primary mission was to produce plutonium and other components for nuclear weapons. Currently, activities at RFETS include cleaning up contamination and waste from its past activities and transitioning its facilities for alternative uses.

- Savannah River Site. The Savannah River Site (SRS) is located in west-central South Carolina and comprises approximately 310 square miles; its production facilities occupy less than 10 percent of the total area. SRS was established by the Atomic Energy Commission in 1950 for the purpose of producing plutonium (Pu-239) and tritium for nuclear weapons. SRS also produced other special isotopes (Cf-252, Pu-238, and Am-241) to support research in nuclear medicine, space exploration, and commercial applications. To produce these radioisotopes, metal targets were irradiated at the five production reactors. Isotopes were recovered from irradiated targets at chemical separation facilities also located at the SRS. Production reactors have operated during different time periods. Current operation of chemical process facilities is limited to the recycling of tritium and the extraction of Pu-238 used in space exploration.

- Weldon Spring. The Weldon Spring Site consists of 229 acres, approximately 20 miles west of St. Louis, Missouri. The Weldon Spring Chemical Plant and the Weldon Spring

Quarry occupy the site. It was part of a site used by the U.S. Army as an ordnance works in the 1940s. In the 1950s and 1960s, the Atomic Energy Commission used the site to process uranium ore in the Weldon Spring Chemical Plant. The plant was subsequently deactivated and no further activities were carried out at the Weldon Spring Site since remediation began in 1985.

Relevant data contained in these four documents are briefly summarized below. Estimates of scrap metal quantities and limited qualitative data are defined in terms of (1) existing scrap metal inventories and (2) projected scrap metal inventories associated with future decommissioning of DOE facilities.

Because significant gaps in quantitative information remain, an attempt was made to supplement reported data by direct contact with DOE personnel. Individuals contacted included key administrative staffs at DOE Headquarters and DOE Regional Offices, as well as persons in DOE field offices. Field personnel included individuals with responsibilities related to scrap metal decontamination, segregation, storage, environmental monitoring, and salvage and recycling operations. In most instances, direct contacts yielded only subjective information that explained the approach and general methods used to arrive at the reported quantities of scrap metal.

4.1.2 Existing Scrap Inventories at DOE

Data Reported in 1996 MIN Report

DOE's first major undertaking to evaluate its materials management practices dates back to January 1990 with the establishment of the Mixed Waste and Materials Management Workgroup. To support the Workgroup effort, an attempt was made to define and inventory Materials Not Classified As Waste (MNCAW) and resulted in the 1994 MIN Report (formerly known as the MNCAW Report). This and other reports have been combined and collated with new data and analysis to provide information presented in the 1996 MIN Report.

DOE defines "materials in inventory" as materials that are not currently in use (i.e., have not been used during the past year and are not expected to be used within the coming year) and that have not been set aside for national defense purposes. The Department identified 10 material categories with significant quantities. The ten categories are divided into two subcategories; scrap metal and equipment is cited among non-nuclear materials (Table 4-1).

Table 4-1. Groupings in DOE Materials in Inventory

Nuclear Materials	Non-Nuclear Materials
○ Spent Nuclear Fuel	○ Sodium
○ Plutonium and Other NMMSS-tracked Materials	○ Lead
○ Natural and Enriched Uranium	○ Chemicals
○ Depleted Uranium	○ Weapons Components
○ Lithium	○ Scrap Metal and Equipment

Scrap metal comprises worn or superfluous metal parts or pieces, including but not limited to structural steel and other metals from decommissioned buildings, scrap metals accumulated from facility maintenance and renovation in the past, and scrap stored in scrap yards and lay-down yards. Scrap metal includes metals that are clean and metals contaminated or activated with radioactivity and/or contaminated with hazardous substances. Equipment considered in the MIN Report is defined as all equipment used for construction, production, or manufacturing and all associated spare parts and hand tools.

To estimate scrap material inventories, the Department recruited personnel from each DOE Operations and Field Office and from designated Headquarters Offices. The MIN Scrap Metal and Equipment Team sought information by means of site-specific surveys and whenever possible extracted information contained in various DOE databases. MIN data collection was, therefore, constrained by the need to use existing data sources with no authorization or resources for new studies and new information. The report acknowledges its limitations and states:

> "... Because of limited data, this report does <u>not</u> attempt to capture the exact amount of each material in inventory. Rather, it attempts to capture the <u>general</u> magnitude of the inventory of each material." (Emphasis added)

Data Reported. Despite acknowledged limitations, the 1996 MIN report is regarded as the principal data source for scrap metal estimates for 10 of the 13 DOE facilities (Table 4-2). Estimates for several facilities presented in Table 4-2 were not directly extracted from the 1996 MIN Report but represent interpolated values.

Table 4-2. Existing Contaminated Scrap Inventories at DOE Sites (Source: 1996 MIN Report)

DOE Site Name	Quantity (metric tonnes)
Fernald	4,218
Hanford	377
Idaho National Engineering Laboratory (INEL)	727
Los Alamos National Laboratory (LANL)	Not Reported
Nevada Test Site (NTS)	264
Oak Ridge National Laboratory (ORNL)	1,129
Y-12	9,065
K-25	29,357
Paducah	48,374
Portsmouth	8,914
Rocky Flats	Not Reported
SRS	13,183
Weldon Spring	Not Reported
Total	115,508

The need for interpolation was due to the fact that only a few DOE sites provided complete quantitative estimates that defined existing scrap metal inventories as clean or radiologically contaminated. Many facilities either provided only a partial breakdown or no breakdown with regard to fractional quantities of contaminated versus uncontaminated scrap metals. In fact, the largest percentage of DOE scrap metal (~80%) as reported in the 1996 MIN Report is designated as "unspecified" with regard to radioactive contamination. For scrap metal inventories designated as "unspecified," it was assumed that 88% of scrap metal was contaminated and 12% was clean and not considered contaminated. The basis for this assumption is defined in Table 1-6, page 16 of Volume 2 of the 1996 MIN Report.

Of existing scrap stockpiles, Table 4-2 identifies that about 90 percent is currently located at five sites. In descending order, they include: Paducah, the K-25 facility in Oak Ridge, the Savannah River Site (SRS), the Y-12 facility in Oak Ridge, and Fernald.

Data Extracted from the 1995 HAZWRAP Report

In 1994, Martin Marietta Energy Systems, Inc., in support of the DOE's Hazardous Waste Remedial Actions Program (HAZWRAP), conducted a study that assessed scrap metal inventories and their economic values for 11 DOE facilities. Collection of information on amounts and locations of scrap metal within the DOE complex was pursued through three independent but complementary methods.

A preliminary questionnaire was forwarded to key site personnel, which requested generic demographic data pertaining to scrap metal management along with a DOE Scrap Metal Data Sheet (Exhibit 4-1). Key information sought by the questionnaire included (1) scrap metal type (e.g., steel, aluminum, copper, etc.), (2) contamination status, and (3) scrap quantities.

A second source of information for developing estimates reported in the HAZWRAP Report came from a thorough review of published reports and DOE databases. A total of 28 documents were identified as pertinent.

Lastly, the Project Team visited the sites and met with personnel to examine storage areas and document the locations and amounts of stored scrap metal. Confirmatory estimates of stored scrap metal quantities were based on physical measurements of size and storage density of piles.

EXHIBIT 4-1

DOE SCRAP METAL DATA SHEET

D.O.E Site ______________________ Date ______________________

Location ______________________ Performed By ______________________

Description ______________________

Material Type	Symbol	Density #/cu.ft.	Density vs Steel %
Aluminum	Al	169	0.34
Copper	Cu	555	1.13
Monel	M	550	1.12
Zirconium	Z	796	1.62
Lead	Pb	710	1.45
Stainless Steel	SS	491	1.0
Steel	Fe	491	1.0
Nickel	Ni	556	1.13

Preparation State	Symbol
Prepared 3x2x5	P
Unprepared	U
Mixed	M
Insulated	Is
25 Box	B
Drum	D
Ingot	Ig
Sealand	Sl
Sheet	S

Radioactive Contamination ___ Uncontaminated ___ Unknown ___

Material Type	Preparation State	Radioactivity	Estimated Volume Yds.	Scrap Density Ton/Yd.	Amount Tons	Comments

Material Type	Preparation State	Radioactivity	Number Units	Unit Weight (Tons)	Amount Tons	Comments

Preparation Equipment (shears, shredders, etc.):

Ongoing Activities:

Comments:

Scrap metal estimates reported in the 1995 HAZWRAP Report were either directly adopted or updated and used in the 1996 MIN Report. As indicated in Table 4-2, scrap metal data for LANL, Rocky Flats, and the Weldon Spring facilities were not fully discussed in the 1996 MIN Report. A brief description of the management and current inventories of scrap metals at these three sites, as reported in the HAZWRAP Report, is presented below.

LANL. Located in New Mexico, the DOE facility has an active scrap metal recycling program. Existing scrap metal inventories are stored at several locations in small piles, the largest of which is about 1,800 metric tonnes. The total quantity of contaminated scrap metal at LANL is estimated at 3,099 metric tonnes.

Rocky Flats. At Rocky Flats, contaminated scrap metal is stored in metal drums and boxes that were inventoried in the Site Waste Management database. Because the material quantities could not be examined using the methods previously described, information from the Waste Management Program was used to quantify amounts and metal types of scrap inventories. A total of 24,543 metric tonnes of contaminated scrap metal was estimated.

Weldon Spring. At the Weldon Spring site in Missouri, scrap metal is located in two storage areas. Contaminated metal scraps removed in the past from process piping associated with the Quarry and Chemical Plants are stored in the Temporary Storage Area (TSA) and in an eight-acre lay-down area called the Material Storage Area (MSA). A total of 27,839 metric tonnes of contaminated scrap metal was estimated.

4.1.3 Summary of Existing Scrap Inventories at DOE Sites

Table 4-3 summarizes current best estimates of contaminated scrap metal quantities currently stored at 13 DOE facilities. Of these estimates, ten were derived from data presented in the 1996 MIN Report. The remaining three values were derived from information presented in the 1995 HAZWRAP Report.

Based on the most current data, it is estimated that existing inventories of scrap metal represent about 171,000 metric tonnes.

Table 4-3. Summary Estimates of Existing DOE Scrap Metal Inventories (Metric Tonnes)

DOE Site	Existing Scrap Metal Quantities	
	MIN Report	HAZWRAP Report
Fernald	4,218	
Hanford	377	
INEL	727	
LANL	Not Reported	3,099
NTS	264	
ORNL	1,129	
Y-12	9,065	
K-25	29,357	
Paducah	48,374	
Portsmouth	8,914	
Rocky Flats	Not Reported	24,543
SRS	13,183	
Weldon Spring	Not Reported	27,839
Subtotal	115,608	55,481
TOTAL	171,089	

4.1.4 Scrap Metal Inventory by Metal Type

Data collected in support of the 1995 HAZWRAP Report provided information regarding the composition of scrap metal inventories. Quantity estimates were provided for seven forms of scrap metal classified as: (1) carbon steel, (2) stainless steel, (3) copper and brass, (4) nickel, (5) aluminum, (6) tin and iron, and (7) miscellaneous, which included lead, monel, and cast iron. These data were reviewed and updated by the MIN Scrap Metal and Equipment Team. Table 4-4 summarizes data reported in the 1996 MIN Report.

Table 4-4. MIN Scrap Metal Inventory by Metal Type (Metric Tonnes)

Metal Type	Clean	Contaminated	Unspecified	Total
Carbon Steel	1,008	11,437	94,472	106,917 (77.2%)
Nickel	0	0	8,817	8,817 (6.4%)
Stainless Steel	1,435	5,392	959	7,786 (5.6%)
Aluminum	27	14	5,637	5,678 (4.1%)
Copper and Brass	24	1,483	147	1,654 (1.2%)
Tin and Iron	227	0	0	227 (0.2%)
Miscellaneous	782	6,537	10	7,330 (5.3%)
Total (percent)	3,503 (2.5%)	24,863 (18.0%)	110,042 (79.5%)	138,409 (~100)

Inspection of Table 4-4 identifies the fact that 3,503 metric tonnes of scrap metal were found to be free of radioactive contamination. Moreover, an estimated 110,042 metric tonnes or about 79.5 percent of existing scrap had not been assessed for radioactive contamination and was classified as "undetermined."

To estimate the fractional quantity of "undetermined scrap" that can reasonably be assumed contaminated and which must be added to the known quantity of contaminated scrap for a total estimated value of contaminated scrap quantity, the following formula was used:

$$\textit{Total contaminated} = (\textit{contaminated scrap}) + \left(\frac{\textit{contaminated}}{\textit{contaminated} + \textit{clean}} \times \textit{undetermined}\right)$$

Table 4-5 provides best estimated values of existing scrap metal inventories that can reasonably be assumed contaminated. Carbon steel comprises more than 77 percent of total tonnage, with nickel, stainless steel, and aluminum representing the other major sources.

Table 4-5. Estimated Scrap Inventories by Metal Type Currently Stored at DOE Facilities (Metric Tonnes)

Metal Type	Contaminated	Assumed Contaminated	Total
Carbon Steel	11,437	125,537	136,974
Nickel	0	11,716	11,716
Stainless Steel	5,392	1,273	6,665
Aluminum	14	7,490	7,504
Copper and Brass	1,483	196	1,679
Miscellaneous	6,537	14	6,551
Total	24,863	146,226	171,089

4.1.5 Future Scrap Metal Quantities at DOE

During peak periods of activities, the nuclear weapons complex represented more than 120 million square feet of building structures (DOE 95a). These buildings include 14 large production reactors with extensive support structures, research reactors and their associated support structures, eight chemical processing plants (i.e., "canyons") with vast quantities of metal piping, tanks, valves, motors, duct-work, and structural components, and an array of buildings used for storage, milling, manufacturing, testing, assembly, and administrative activities.

With the end of the Cold War Era and the questionable need for additional nuclear weapons, many of these structures will be decommissioned over the next several decades. As of June 1995, DOE's Office of Environmental Restoration Decommissioning Inventory slated 865 structures for future decommissioning (U.S. DOE, Office of Environmental Restoration Decommissioning Inventory, June 1995).

Several facilities are still awaiting final notification for deactivation and are not yet designated for decommissioning. As a result, assessments aimed at estimating future scrap generation at some DOE sites have not been conducted for these facilities.

Site-Specific Estimates

For those DOE sites that have been identified for partial or total decommissioning, scrap quantities are at best preliminary estimates that are based on limited and incomplete data. Projected scrap estimates associated with future decommissioning activities were derived from three reports that include the following facilities:

- <u>EPA 95 Report</u>
  Sites: Fernald, Hanford, LANL, Rocky Flats
- <u>MIN 96 Report</u>
  Sites: INEL, SRS
- <u>DOE 95 Report</u>
  Sites: K-25, Paducah, Portsmouth

Combined scrap quantities from future decommissioning activities at these sites are estimated at around 925,000 metric tonnes. Scrap sources and site-specific estimates for the ten facilities are briefly defined below.

<u>Hanford</u>. To date, only modest attempts have been made to assess future scrap quantities pertaining to decommissioning activities. However, quantities are expected to be substantial. As of June 1995, 250 buildings at Hanford had been slated for decommissioning. Massive amounts of structural steel scrap will be produced during the decommissioning of these buildings. Also included are other structures such as exhaust stacks, storage tanks, and river outfall structures as well as carbon steel and stainless steel pressure vessels from the Clinch River Breeder Reactor program.

Approximately 91,798 metric tonnes of scrap are likely to be generated during decommissioning activities. The vast majority of scrap is expected to be carbon steel with significant amounts of stainless steel, lead, and aluminum.

<u>Idaho National Engineering Laboratory</u>. Over the past 50 years, more than 50 nuclear reactors (mostly small test reactors) have operated at INEL. While some of these reactors and their support buildings have already undergone decommissioning, others are targeted for future decommissioning. Many published DOE documents that cite scrap estimates were assessed in

EPA 95 and in the MIN 96 Report. Future decommissioning activities at INEL are estimated to generate 33,486 metric tonnes of scrap metal. At this facility, carbon steel (55.7%) and stainless steel (44.0%) represent nearly all projected scrap metal quantities.

Los Alamos National Laboratory. LANL's Metal Inventory Report (INV 96) not only assessed existing scrap metal inventories but identified future scrap metal quantities associated with decommissioning activities, as well as for scheduled "upgrade" projects. In combination, decommissioning and upgrade activities are estimated to generate 2,686 metric tonnes of scrap.

Fernald. Fernald's production area includes 20 process facilities and supporting structures that are obsolete and beyond their design life. In total, 128 buildings and 72 miscellaneous structures have been identified for decontamination and decommissioning. The dismantling of buildings, process equipment, and structures is estimated to generate 135,623 metric tonnes of scrap.

Savannah River Site. This site includes five heavy water production reactors that were used in the production of tritium and other weapon materials. All reactors have been shutdown and are likely to be disassembled. Scrap associated with the decommissioning of the five production reactors and support structures/systems is estimated at 3,054 metric tonnes with nearly equal contributions of carbon steel and stainless steel. The fate of the two SRS chemical separation plants and the many facilities that support them remains undetermined. The decommissioning of these facilities would undoubtedly add substantial (but to date undefined) quantities of scrap.

Rocky Flats. A literature search in support of the EPA95 revealed the existence of only one study that estimated future scrap quantities for Rocky Flats. A study by the Manufacturing Sciences Corporation (MSC 94) stated that the decommissioning of Rocky Flats is expected to generate about 1,003 metric tonnes of scrap metal from four buildings that will be used for the National Conversion Pilot Project and an additional 25,300 metric tonnes from the other buildings and site structures. Most scrap is likely to be contaminated with depleted uranium, enriched uranium, and/or plutonium.

Oak Ridge, K-25 Facility. The K-25 facility is the first of three DOE gaseous diffusion plants that are slated for decommissioning. Decommissioning of the K-25 site is estimated to take a total of eleven years: two years of planning and nine years of decontamination and decommissioning. Current projections are that decommissioning activities will be completed in the year 2006.

Decommissioning will include removal of large quantities of metals associated with process equipment, piping, and structural components. Principal contaminants include uranium and daughter products, technetium-99, and trace quantities of neptunium-237 and plutonium-239. DOE 95 identified a total quantity of 406,372-metric tonnes of recyclable metal but did not identify the fractions of uncontaminated and contaminated scrap metal.

Personal communications with Gary Person (PERS 96), principal author of DOE 95 Report, yielded the following estimates: of the total future inventory of 406,273 metric tonnes of scrap metal, 193,666 are estimated to be free of contamination and about 212,706 are likely to be moderately contaminated scrap that is considered suitable for recycling.

Portsmouth. Decommissioning of the Portsmouth gaseous diffusion facility is scheduled to begin in FY 2007 (following completion of decontamination and decommissioning activities at the K-25 facility), with a completion date in FY 2015. The decontamination and decommissioning of the three gaseous diffusion plants are purposely scheduled in series in order to (1) learn from experience gained, (2) minimize yearly expenditures, and (3) provide a steady stream of metal for recycle. DOE 95b reported the availability of 312,085 metric tonnes of total scrap metal. Of this quantity, 189,072 metric tonnes are estimated to be contaminated metal that, with decontamination, is considered suitable for recycling.

Paducah. The Paducah Gaseous Diffusion Plant is the third and last gaseous diffusion facility to be decommissioned. Current projections identify decommissioning to start in 2015 and end in 2023. The first major phase will be the removal and decontamination of major components (i.e., motors, cell housing, compressors, compressors and converters, piping and valves, electrical equipment, and HVAC system) from the process buildings. Personal communication (PERS 96) identified that of the total projected scrap metal inventory of 331,365 metric tonnes (DOE 95b) about 230,886 are estimated to represent scrap that is considered suitable for recycling.

4.1.6 Summary and Conclusions

At its peak, the nuclear weapons complex consisted of 16 major facilities that represented more than 120 million square feet of buildings. These buildings contain large quantities of equipment, structural steel, and other metal components. Over the 50-year period, some of these buildings, their ancillary facilities, and the equipment they housed have been renovated, replaced, and/or demolished. Currently, more than 170,000 metric tonnes of low-level contaminated scrap metal is stored at various facilities that are considered suitable for recycling.

Estimates of existing scrap quantities are mostly based on site-specific review of historical inventory data and physical surveys of scrap piles. Quantity estimates of existing scrap inventories can, therefore, be viewed with reasonable confidence.

Future scrap quantities are closely linked to projected decommissioning activities at DOE sites that make up the nuclear weapons complex. At some sites, virtually all structures and their contents will be dismantled and removed; at other sites decommissioning may be limited, and the DOE will continue select operations considered crucial to national security or important to national research. To date, decisions and commitments for decommissioning are not only incomplete but, in instances where such decisions have been made, they remain both tentative and are subject to change in scope and schedule. Consequently, estimates of future scrap quantities are less certain.

In this report, future scrap estimates were based on currently scheduled decommissioning activities at nine facilities that include Fernald, Hanford, INEL, LANL, SRS, Paducah, Y-12 and K-12. Decommissioning of these facilities is estimated to yield more than 925,000 metric tonnes of contaminated scrap metal that is derived from dismantling large production reactors, research reactors, chemical processing plants, and a vast array of associated support facilities and structures. With effective decontamination, this scrap metal is potentially available for recycling.

Table 4-6 provides summary estimates that represent existing scrap inventories and future scrap associated with decommissioning activities. Of the more than one million metric tonnes of scrap, about 85 percent represents carbon steel with near equal quantities of copper, nickel, aluminum, and stainless steel representing the remainder. It is possible that these values may underestimate the total scrap metal quantities due to the fact that current data pertaining to future decommissioning activities are incomplete.

Table 4-6. Summary Data for Existing and Future Contaminated Scrap at DOE Facilities*
(Metric Tonnes)

Site Name	Scrap Metal Database Volume	Metal Type													
		Al	C. Steel	S. Steel	Copper	Nickel	Monel	Lead	Cast Iron	Black Iron	Graphite	Cu/Brass	Tin/Fe	Other	Misc.
Fernald	139,841	---	101,753	---	38,088	---	---	---	---	---	---	---	---	---	---
Hanford	92,175	684	87,020	787	---	24	---	291	---	---	1,632	5	1	1,711	20
Idaho	34,213	30	19,195	14,733	44	44	---	110	4	7	---	8	2	36	---
LANL	5,785	40	5,568	177	---	---	---	---	---	---	---	---	---	---	---
NTS	264	11	204	15	---	17	---	---	---	---	---	2	1	---	14
ORNL	1,129	18	992	117	2	---	---	---	---	---	---	---	---	---	---
Y-12	9,065	33	8,392	602	38	---	---	---	---	---	---	---	---	---	---
K-25	242,063	7,988	232,953	753	304	---	65	---	---	---	---	---	---	---	---
Paducah	279,260	21,161	212,917	190	198	44,794	---	---	---	---	---	---	---	---	---
Portsmouth	197,986	6,130	191,412	18	408	---	18	---	---	---	---	---	---	---	---
Rocky Flats	50,846	---	33,666	2,454	14,726	---	---	---	---	---	---	---	---	---	---
SRS	16,237	14	10,403	5,809	11	---	---	---	---	---	---	---	---	---	---
Weldon Sp	27,839	510	26,877	406	46	---	---	---	---	---	---	---	---	---	---
TOTAL	1,096,703	36,619	931,352	26,061	53,865	44,879	83	401	4	7	1,632	15	4	1,747	34
Percent of Total	100.00	3.34	84.92	2.38	4.91	4.09	0.01	0.04	3.6E-6	6.4E-6	0.15	1.4E-5	3.6E-6	0.16	3.1E-5

* Includes metal for which decisions regarding its disposition may be affected by an EPA recycling standard.

4.2 POTENTIAL SCOURCES AND CHARACTERISTICS OF SCRAP METAL FROM THE COMMERCIAL NUCLEAR POWER INDUSTRY

The U.S. commercial nuclear power industry is represented by 123 reactor plants. At present, eight reactors have been shutdown; in the next two to three decades, most of the reactors currently in operation will have reached their projected forty-year lifetime. A great deal of information and data has been compiled by the U.S. Nuclear Regulatory Commission (NRC) and the individual utilities pertinent to the decommissioning of these facilties and the associated quantities and characteristics of the scrap metal that will be produced. Appendix A presents a detailed summary of the pertinent information. This section summarizes the information provided in Appendix A.

A crucial factor affecting the quantity of scrap metal and associated contamination levels is the basic design of the reactor. The two types of reactors used in the United States are the pressurized water reactor (PWR) and the boiling water reactor (BWR). Of the 123 U.S. reactor units, 40 are BWRs manufactured by General Electric (GE) and 83 are PWRs manufactured by Westinghouse (W), Combustion Engineering (CE), and Babcock and Wilcox (B&W). In the 1976-1980 time frame, two studies were carried out for the U.S. Nuclear Regulatory Commission by the Pacific Northwest Laboratory (PNL) that examined the technology, safety, and costs of decommissioning large reference nuclear power reactor plants. Those studies, NUREG/CR-0130 and NUREG/CR-0672 for a reference PWR and reference BWR, respectively, reflected the industrial and regulatory situation of the time. To support the final Decommissioning Rule issued in 1988, the earlier PNL studies have been updated with the recent issuance of NUREG/CR-5884, Revised Analyses of Decommissioning for the Reference Pressurized Water Reactor Station and NUREG/CR-6174, Revised Analyses of Decommissioning for the Reference Boiling Water Reactor Power Station. These four NUREG reports along with several other NRC reports and select decommissioning plans on file with the Commission represent the primary source of information used to characterize Reference PWR and BWR facilities and to derive estimates of scrap metal inventories for the industry at large.

4.2.1 Summary Estimates of Contaminated Steel for Reference BWR/PWR and the Commercial Nuclear Industry

Table 4-7 presents summary data for contaminated steel potentially available for recycling. Estimates for the Reference BWR and PWR were derived by summing component mass values previously cited in Table A5-2 and Table A5-3, respectively, of Appendix A. Estimates for the entire commercial nuclear industry were derived by taking Reference BWR and Reference PWR values and applying plant-specific scaling factors for each of the 40 BWRs and 83 PWRs. Approximately 600,000 metric tonnes of contaminated steel may become available over time for recycling. About 80% of the contaminated steel is carbon steel, with stainless steel representing the balance.

Table 4-7. Summary Data for Contaminated Steel Inventories Potentially Suitable for Recycling

Contaminated Material(a)	Quantity (metric tonnes)				
	Reference BWR(b)	Reference PWR(c)	All BWRs(b)	All PWRs(c)	Total Industry
Stainless Steel	1,688	827	56,987	64,738	121,725
• Low-level Contamination ($<1 \times 10^5$ dpm/100 cm^2)	576	210	19,446	16,439	35,885
• Medium-level Contamination (1×10^5 to 1×10^7 dpm/100 cm^2)	786	114	26,535	8,924	35,459
• High-level Contamination ($>1 \times 10^7$ dpm/100 cm^2)	326	503	11,006	39,375	50,381
Carbon Steel	6,754	3,311	228,015	259,185	487,200
• Low-level Contamination ($<1 \times 10^5$ dpm/100 cm^2)	2,306	841	77,851	65,834	143,685
• Medium-level Contamination (1×10^5 to 1×10^7 dpm/100 cm^2)	3,146	458	106,209	35,852	142,061
• High-level Contamination ($>1 \times 10^7$ dpm/100 cm^2)	1,302	2,012	43,955	157,499	201,454
TOTALS	8,442	4,138	285,002	323,923	608,925

(a) Although data for stainless steel and carbon steel are presented as independent quantities, it must be acknowledged that a significant fraction of stainless steel is unlikely to be segregated as such for recycling purposes.

(b) BWR radionuclide contaminants reflect those of Table A3-11 and Table A3-21 of Appendix A.

(c) PWR radionuclide contaminants reflect those of Table A3-17 and Table A3-20 of Appendix A.

Because past and current regulatory release criteria (i.e., U.S. NRC Regulatory Guide 1.86; NRC 74) are defined in activity levels per unit surface area, information cited in this section has been presented in this fashion. However, for risk analysis pertaining to recycling of scrap metals, a more meaningful approach is to express contamination levels in terms of activity per unit mass. This conversion requires the derivation of the average mass thickness (g/cm²) of metal surfaces by the following equation:

$$Average\ Mass\ Thickness\ (g/cm^2) = \frac{\sum Surface\ Areas\ (cm^2)}{\sum Metal\ Mass\ (g)}$$

For the contaminated systems/components previously identified for Reference BWR and PWR, a weighted average mass density of 3.5 g/cm² for contaminated surfaces was estimated. At a density of about 8 g/cm³ for steel, this corresponds to an average thickness of about 4.4 mm (0.17 inches). This average mass thickness can now be readily applied to estimate the activity level per unit mass of contaminated steel. For example, under the current interim release criteria of 5,000 dpm/100 cm² for beta-gamma emitters, the residual contamination on average would correspond to about 14 dpm/g (or about 6.5 pCi/g.; or 0.23 Bq/g) of steel.

4.2.2 Contaminated Metal Inventories Other Than Steel

There are significant quantities of metals and metal alloys other than steel that may be suitable for recycling, including: (1) galvanized iron, (2) copper, (3) inconel, (4) lead, (5) bronze, (6) aluminum, (7) brass, (8) nickel, and (9) silver. However, there exist no credible data in the open literature regarding the estimated fraction(s) of these metal inventories that are likely to be contaminated or the extent of their contamination. In the absence of reported data, a reasonable approach may assume that the contaminated fraction among total plant inventories of these metals parallels the contaminated fraction of carbon steel for Reference BWR and Reference PWR. Justification for this modeling approach is based on the fact that most of these metals exist as sub-components of larger items consisting primarily of carbon steel. From data cited in Appendix A, the percent of contaminated carbon steel suitable for recycling to that of total plant inventory corresponds to 20% and 10% for Reference BWR and Reference PWR, respectively. The application of these values yields the contaminated metal quantities (suitable for recycling) cited in Table 4-8. Due to physical differences and chemical properties that affect corrosion and internal contamination, categorization of the contamination of these metals using the methods used for steel is not appropriate.

Table 4-8. Summary of Contaminated Metal Quantities Other than Steel
(metric tonnes)

Metal Type	Reference Facility		Industry		
	BWR	PWR	All BWRs	All PWRs	Total
Galvanized Iron	258	130	8,710	10,037	18,747
Copper	137	69	4,625	5,327	9,952
Inconel	24	12	810	927	1,737
Lead	9.1	4.6	307	355	662
Bronze	5.0	2.5	169	193	362
Aluminum	3.6	1.8	122	139	261
Brass	2.0	1.0	68	77	145
Nickel	0.2	0.1	7	8	15
Silver	<0.2	<0.1	<7	<8	<15

4.2.3 Time-Table for the Availability of Scrap Metal from the Decommissioning of Nuclear Power Plants

For currently operating nuclear power plants, an operational period of 40 years is assumed. Following reactor shutdown, a minimum of 10 years is assumed before significant quantities of scrap metal would be available. Thus, for currently operating reactors, the earliest dates for releasing scrap metal are defined by their startup dates plus 50 years. Currently, there are eight reactor units that have been permanently shutdown (Dresden-1 (1984); Indian Point-1 (1980); LaCrosse (1981); TMI-2 (1979); Humboldt Bay (1976); Trojan (1993); Rancho Seco (1989); San Onofre-1 (1992); and Yankee Rowe (1992)). A conservative assumption for these facilities projects the release of scrap metal over a ten-year period between 2000 and 2009.

Table 4-9 summarizes the potential availability of scrap metal for yearly intervals starting with the year 2010. The release of scrap metal based on this time-table should, however, be considered highly conservative since many, if not most, facilities are likely to delay decommissioning activities for varying portions of the allowable 50-year SAFSTOR period.

Table 4-9. Time-Table for Available Contaminated Scrap Metals from Decommissioned Nuclear Power Plants
Quantities (metric tonnes)

Year	CS	SS	Galv. Iron	Copper	Inconel	Lead	Bronze	Alum.	Brass	Nickel
2000-2009	15,377	4,107	609	323	57	21	12	8	5	0.5
2010	804	105	31	17	3	1	<1	<1	<1	—
2011	—	—	—	—	—	—	—	—	—	—
2012	—	—	—	—	—	—	—	—	—	—
2013	—	—	—	—	—	—	—	—	—	—
2014	3,616	475	140	75	13	5	3	2	1	—
2015	3,616	475	140	75	13	5	3	2	1	—
2016	—	—	—	—	—	—	—	—	—	—
2017	6,464	1,958	257	136	24	9	5	4	2	0.2
2018	14,811	2,712	580	308	54	20	11	8	4	0.5
2019	1,763	534	70	37	7	2	1	1	<1	0.1
2020	15,442	2,740	604	321	56	21	12	8	4	0.5
2021	2,438	739	97	51	9	3	2	1	<1	0.1
2022	10,328	1,674	370	197	34	13	7	5	3	0.3
2023	38,415	8,571	1,512	439	141	53	29	21	12	1.2
2024	41,117	8,855	1,616	859	151	57	31	22	12	1.3
2025	12,927	3,175	510	271	48	18	10	7	4	0.4
2026	22,927	5,326	903	480	84	31	17	12	7	0.7
2027	5,611	1,700	223	118	21	8	4	3	2	0.2
2028	9,574	2,185	377	200	35	13	7	5	3	0.3
2029	—	—	—	—	—	—	—	—	—	—
2030	9,078	2,750	362	192	34	13	7	5	3	0.3
2031	8,961	2,714	357	189	34	12	7	5	3	0.3
2032	10,697	1,406	415	221	38	15	8	6	3	0.3
2033	11,191	2,468	440	234	41	16	8	6	3	0.3
2034	30,466	5,672	1,193	635	111	42	23	16	9	1.0
2035	26,173	6,206	1,032	548	96	36	20	14	8	0.8
2036	32,396	6,248	1,269	675	118	45	24	17	10	1.0
2037	13,192	3,996	525	278	49	18	10	7	4	0.4
2038	9,637	2,919	384	203	36	13	7	5	3	0.3
2039	8,366	1,639	328	174	30	12	6	4	2	0.2

Table 4-9. Time-Table for Available Contaminated Scrap Metals from Decommissioned Nuclear Power Plants (Continued)
Quantities (metric tonnes)

Year	CS	SS	Galv. Iron	Copper	Inconel	Lead	Bronze	Alum.	Brass	Nickel
2040	12,956	3,925	516	273	48	18	10	7	4	0.4
2041	—	—	—	—	—	—	—	—	—	—
2042	3,261	988	130	69	12	5	2	2	1	0.1
2043	—	—	—	—	—	—	—	—	—	—
2044	—	—	—	—	—	—	—	—	—	—
2045	2,703	819	107	57	10	4	2	1	1	0.1
2046+	12,868	3,902	512	271	48	18	10	7	4	0.4
Total	397,175	90,983	15,609	7,926	1,455	547	299	212	121	12.2

REFERENCES

DOE 95a	"Closing the Circle on the Splitting of the Atom," U.S. Department of Energy, Office of Environmental Management, January 1995.
DOE 95b	"Gaseous Diffusion Facilities Decontamination and Decommissioning Estimate Report," prepared by G.A. Person, et al, Environmental Restoration Division, Oak Ridge, TN for U.S. DOE, Office of Environmental Management, ES/ER/TM-171, December 1995.
EPA/SCA 95	"Analysis of the Potential Recycling of Department of Energy Radioactive Scrap Metal" prepared by S. Cohen & Associates, Inc. for the U.S. Environmental Protection Agency, Office of Radiation and Indoor Air, August 1995.
HAZ 95	"U.S. Department of Energy Scrap Metal Inventory Report for the Office of Technology Development, Office of Environmental Management," prepared by Hazardous Waste Remedial Actions Program for the Department of Energy, DOE/HWP-167, March 1995.
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PERS 96 Personal communications with G.A. Person, principal author of the "Gaseous Diffusion Facilities Decontamination and Decommissioning Estimate Report," Lockheed Martrin Energy Systems, Inc., October 26, 1996.

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CHAPTER 5

DESCRIPTION OF UNRESTRICTED RECYCLING OPERATIONS

5.1 INTRODUCTION

5.1.1 Recycling Scrap Steel—An Overview

Figure 5-1 presents a simplified schematic diagram of the steps that would be involved in recycling carbon steel scrap into consumer or industrial products. The process starts with steel scrap that already exists in scrap piles at various DOE and perhaps at NRC-licensed facilities, or that will be generated in the course of the decontamination and decommissioning (D&D) of such facilities. Scrap that has been cleared for release is loaded onto trucks to be transported off site. As indicated in Figure 5-1, any required decontamination takes place within a radiation controlled area (RAC). All operations within the RAC are performed by occupationally exposed workers, who are subject to DOE- or NRC-regulated exposure limits and ALARA procedures. Therefore, the exposures of these workers are not considered in the present analysis.[1]

The scrap is transported to a processor where it is unloaded, sorted and possibly cut up or compacted. The processed scrap is transported to a steel mill where it may be unloaded to a scrap pile or sent directly to the furnace. In either case, it is loaded into a charging bucket and charged to an electric arc furnace (EAF), where it is melted.

Certain constituents of the furnace charge are either volatilized or entrained in the air stream as particulate matter. Most of these emissions are captured by the emission control system and routed to the baghouse, where the fumes are cooled and filtered. The filters, which are in the form of long bags, are periodically emptied by remotely operated mechanical means. The dust is transferred to a tanker truck and shipped off site.

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[1] Operations indicated by cross-hatched boxes in the diagram are not modeled in the present analysis.

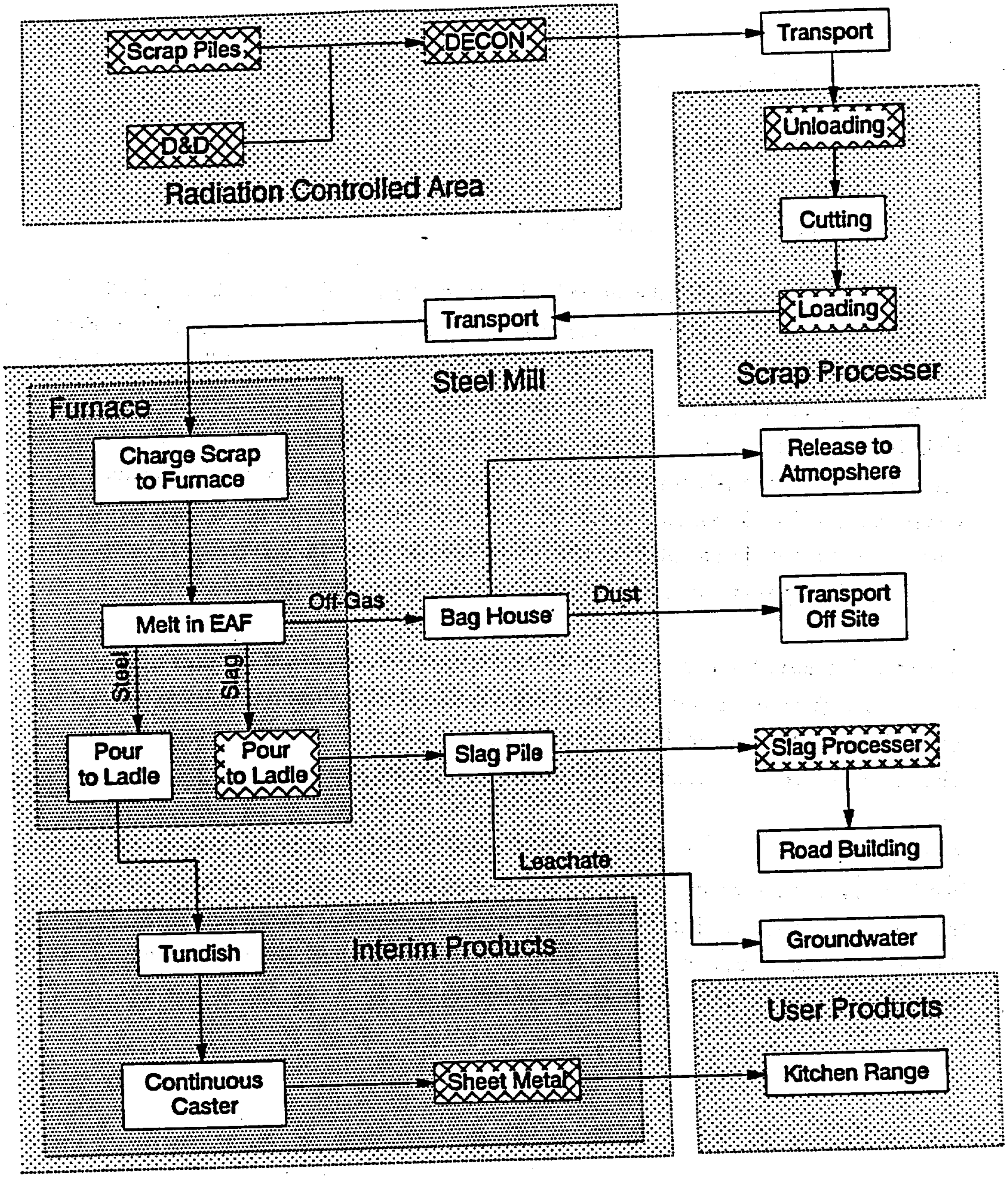

Figure 5-1. Operations Analyzed

After the scrap is melted, first the slag and then the molten steel are poured into ladles. The molten steel is transferred from the ladle to a tundish from which it is fed to a continuous caster, where it is made into slabs. These may be sold as such or made into interim mill products, such as coils of sheet metal.

The slag is transported to a slag pile at the steel mill, where it is stored prior to shipment to a slag processing facility. The slag processor sells the slag for various uses, such as ballast for road-building or aggregate which is mixed with cement and used for paving. While the slag is stored at the mill, various components may leach out and percolate through the soil to an underlying aquifer, possibly contaminating an underground source of drinking water.

5.1.2 Reference Facility

In the United States, most steel scrap is melted in either an EAF or a basic oxygen furnace (BOF). The charge for an EAF usually consists entirely of scrap, while scrap makes up less than 30% of the feedstock of a BOF, the rest being the pig iron output of a blast furnace. It is therefore possible for a given EAF charge to consist exclusively of scrap from nuclear facilities, while in the case of a BOF, the scrap would be diluted by pig iron. A steel mill equipped with EAFs was therefore selected as the reference mill for the present study.

It is unlikely, however, that, for an entire year,[2] any steel mill would be exclusively supplied with scrap resulting from the dismantling of components that had been potentially exposed to radioactive contamination. To determine the largest fraction of the scrap that could be potentially contaminated, the anticipated release of scrap metal by various generator sites nationwide was matched to the scrap processing capacities of nearby steel mills. It was found that four nuclear power plants in a single locality are scheduled for decommissioning in the same year. Metal workers would receive the highest likely exposures during that peak year if all the recyclable scrap metal from these plants were taken to a single scrap processor and melted at a small nearby steel mill. A detailed report of the study is found in Appendix G.

The reference steel mill for the present analysis was based partly on the Calumet Steel Co. facility in Chicago Heights, IL, which is described in greater detail in Appendix H. The mill is

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[2] The present analysis assesses the radiological impacts during the year of peak exposure.

equipped with two EAFs, each of which has a 12.5-foot diameter shell and produces a 32-ton average heat, with a nominal capacity of 75,000 tons per year. Other parameters used in the analysis are based on data pertaining to other facilities, on engineering judgment, and on analytical assumptions. Therefore, the results of the present analysis are not intended, and should not be used, to predict the radiological impacts of recycling scrap metal on any real individual or facility, now or in the future.

5.1.3 Exposure Pathways

The exposure pathways considered in the present analysis fall into two general groups: external exposure to direct penetrating radiation and internal exposure from inhaled or ingested radionuclides.

External Exposure

The external exposures are evaluated by use of the MicroShield™ computer code, which is described in more detail in Section 6.3.1, or by dose coefficients adapted from Federal Guidance Report (FGR) No. 12 (Eckerman 93).

MicroShield™ computes the exposure rate from a uniform distribution of one or more radionuclides within a specified matrix, such as a solid cylinder of iron, with additional shielding material between the source (i.e., the iron cylinder) and the receptor point. MicroShield™ includes attenuation and build-up factors for nine metallic elements as well as air, concrete and water. In addition, it is possible for the user to create custom materials by specifying the densities and elemental compositions of the new material. However, the present analysis uses iron to represent the various steel alloys in both the source material being processed and in the components of the furnace that act as radiation shields. Since carbon steel contains over 98% iron, it is preferable to model it as pure iron, as the build-up factors for iron are based on actual measurements.

Examples of the MicroShield™ external exposure assessments of all relevant scenarios are shown in Appendix H.

Internal Exposure

The internal exposure pathways consist of the inhalation of radioactively contaminated dust, the incidental ingestion of contaminated dust, soot or other loose, finely divided material, and the ingestion of contaminated food or water.

The following sections describe the geometries and the materials used to model the external exposure from each task, as well as the assumptions regarding the inhalation and ingestion pathways. A detailed discussion of the last two pathways appears in Sections 6.3.2 and 6.3.3.

5.2 LIST OF OPERATIONS AND EXPOSURE SCENARIOS

Table 5-1 lists the operations and exposure parameters employed in the assessment of potential radiological impacts of recycling scrap metal on the RMEI. These operations and the parameters used to model the corresponding exposure scenarios were based on an earlier EPA study on the recycling of DOE scrap metal (SCA 95). That study comprised over 60 exposure scenarios, which were selected on the basis of reviews of the literature, including work done for the International Atomic Energy Agency (IAEA 91) and the U.S. NRC (O'Donnell 78), as well as observations of current scrap metal recycling practices made during visits to steel mills and scrap processors, and on the basis of data furnished by personnel of these facilities. The present analysis encompasses those operations shown to have the maximum potential impacts on the exposed individuals. The new exposure scenarios utilize information obtained in the course of additional visits to steel mills and scrap processors, discussions with steel industry personnel, further research into scrap metal recycling, and the professional judgments and experience of the project team, whose combined professional experience encompasses health physics, industrial hygiene and metallurgy.

As seen in Table 5-1, the study also assesses the radiation exposures from several representative finished products which might be made from recycled scrap.[3] These products were selected on the basis of their wide use and their potential radiological impacts on individuals and/or

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[3] Three of these products are made from cast iron, which is produced by a different process than is used to make carbon steel. Since the radiological impacts of iron founding are not included in the present study, these products are not represented in Figure 5-1. However, the contaminant distributions characteristic of cast iron are utilized in the impact assessment of these products (see Section 6.2 and Appendix F).

population groups—they are comparable to the finished products in the earlier studies. For many radionuclides, the impacts on end users would be dominated by exposure to external radiation. Therefore, the highest impacts would be produced by massive products that are in close proximity to the exposed individuals for the longest times. Cooking utensils were included to assess radiation exposures from consumption of food potentially contaminated by radionuclides leaching from the metal into the food during cooking.

5.2.1 Dilution Factors

As discussed above, potentially contaminated scrap would in most cases be diluted with scrap that had never been exposed to radioactive contaminants. The ratio of the total amount of metal to the potentially contaminated scrap—termed the dilution factor—is listed for each of the four major groups of operations shown in bold-faced upper-case type in Table 5-1.

Scrap Processing

As was previously stated, all the recyclable scrap metal from four commercial nuclear power plants could plausibly be sent to a single scrap processor in a single year. However, as cited in Appendix G, only 13% of such scrap would be potentially contaminated. Therefore, the average specific activity of any given radionuclide that in the scrap being processed during the course of the year would be only 13% of its average specific activity in the potentially contaminated scrap.[4] The dilution factor for the scrap processing operations is thus:

$$\frac{1}{0.13} = 7.7$$

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[4] Throughout this report, the terms "radionuclide concentration" and "specific activity" may appear to be used interchangeably. Strictly speaking, concentration refers to a given physical quantity, such as mass, per unit volume or unit mass of the matrix. The concentration of uranium in slag, for example, might be expressed in micrograms of uranium per gram of slag. Specific activity is always expressed in units of activity per unit mass, such as pCi/g. For a given radionuclide, of course, the specific activity is proportional to its concentration. Since radionuclides are usually detected and assayed in terms of their activities, not in terms of their masses, specific activity is a more useful concept.

Table 5-1. Operations and Exposure Parameters for Radiological Assessments of Individuals

Description	Mnemonic	Dilution factor	Exposure Pathways						
			External Exposure			Internal			
			Time (hr/y)	Distance	Medium	Time (hr/y)	Medium	Dust load (mg/m^3)	RF[a]
Truck driver transporting scrap	SCRPDRVR	7.7	2000	8 ft	scrap	N/A			
SCRAP PROCESSING									
Cutting scrap	SCRAPCUT	7.7	1750	N/A[b]	scrap	1500		10	0.5
STEEL MILL									
Furnace Operations		9.1							
Crane Operator	OP-CRANE		1750	10 m	scrap	1750	dust[c]	1.3	0.58
EAF furnace operator	FURNACE		1750	4'—30'[d]		1750		2.2	
Airborne effluent emissions	AIRBORNE		N/A						
Interim Products									
Operator of continuous caster	OPCASTER		1750	2'—15'	steel	1750	dust	2.0	0.58
Bag House									
Bag house maintenance	BAGHOUSE		1460	10 m	scrap	1460	dust	2.3	0.58
			40	[e]	dust	40		40	0.076[f]
			240	[e]		240		1.2	0.58
Truck driver: baghouse dust	DUSTDRIV		135	3.5 m		N/A			
Slag Pile									
Slag pile worker	SLAGPILE		1000	N/A[b]	slag	1750	slag	2.6	0.51
Slag leachate in groundwater	SLGLEACH		N/A						
USE OF MILL PRODUCTS									
Using slag in road construction	SLAGROAD	8.7	140	1 m[b]	slag	1750	slag	2.6	0.51
Assembling automobile engines	ENGNWRKR		1750	.2—.7 m	cast Fe	N/A			
Manufacturing industrial lathe	LATHEMFG		1750	.2—.7 m		1750	cast Fe	2.7	0.5
END USERS									
Using kitchen range	COOKRNGE	1.0	525	2 ft	steel	N/A			
Taxi driver	TAXIDRVR		1750	2 ft	cast Fe				
Lathe operator	OP-LATHE		1750	.2—.7 m					
Cooking on cast iron pan	FEFRYPAN		263	2 ft		N/A[g]	cast Fe	N/A	

[a] Respirable Fraction
[b] Exposure assessment uses FGR 12 dose coefficients—see discussion in Section 6.3.1
[c] Dust = baghouse dust
[d] Range of distances—see discussion in Section 6.3.1
[e] Special model—see discussion in Appendix H
[f] Includes respiratory protection factor of 0.1
[g] Exposure to contaminated food

Steel Mill

The two EAFs at the reference steel mill have a total nominal capacity of 150,000 tons per year. However, as discussed in Appendix G, the postulated decommissioning of the four nuclear power plants would only yield a total of 132,000 tons of scrap, of which 13%, or about 17,000 tons, would be potentially contaminated. These 17,000 tons represent approximately 11% of the mill's annual capacity. Thus, the dilution factor for the steel mill operations is $\frac{1}{0.11} = 9.1$.

Use of Mill Products

It was assumed that the three operations using mill products modeled in this analysis obtained all their materials from the reference mill. Thus, the materials are assigned the same dilution factor as the steel mill operations.

End Users

Since any one item could be made from a single heat which could consist only of potentially contaminated scrap, the dilution factor for the end user scenarios is equal to 1.

5.2.2 Scrap Processing Operations

Assessments were performed on two workers involved in processing scrap. One is a truck driver who spends eight hours per day in the cab of a truck carrying 21-ton loads of scrap metal to the scrap processor. His only exposure would be to external radiation from the load of contaminated scrap. Another is a worker who spends six hours per day cutting the scrap, but spends a total of seven hours in canyons surrounded by scrap. He would also inhale and ingest dust which is assumed to have the same specific activity as the scrap.

5.2.3 Steel Mill

Most mills that process scrap metal receive the scrap via truck or rail. Upon arrival at the mill, the scrap is unloaded, charged into an EAF and melted. Steel mills typically minimize the unloading scrap into piles because of the extra cost of reloading the metal for transportation to the furnace.

Furnace Operations

The scrap-bearing container is unloaded by means of a large electromagnet and dumped into charging buckets that move the scrap to the furnace. The exposures of two workers performing representative tasks involved with furnace operations are assessed in the present analysis. One is the crane operator who transfers the charging bucket—he may be exposed to external radiation from the scrap in the bucket. The other is the furnace operator, who may be exposed to radiation from the scrap in the furnace while it is melting. They both may be exposed to fugitive furnace emissions which escape capture by the emission control system. Such emissions may also lead to radiation exposures of the population living near the facility via the following pathways:

- External exposure from immersion in a plume of radioactive emissions
- External exposure from radioactive particles deposited on the ground
- Inhalation of radioactive emissions
- Consumption of vegetables raised on contaminated soil
- Consumption of milk and beef from cattle raised on contaminated forage

Interim Products

Once the steel melts, it is poured either into ingot molds or onto a continuous caster which produces steel slabs. At a continuous caster, torches cut the slabs into smaller pieces as the steel runs down a set of rollers. Cooled slabs are stored, reheated and formed into products such as coils of sheet metal, or are sold as such. The operator of the continuous caster may be exposed to external radiation from the molten steel in the tundish as well as from the slabs cast in the continuous caster. He may also be exposed to fugitive furnace emissions.

Baghouse

The baghouse contains rows of filters, suspended from the ceiling, that trap the various effluent emissions from the melt-refining process. These bag-like filters are continuously shaken; the dust settles into collecting hoppers and is fed by a screw mechanism into a tanker trailer. Once filled to capacity, the trailer is transported away from the steel mill to a processing facility for recovery of commercially valuable components, primarily metals such as zinc, cadmium and lead, and for ultimate disposal.

Steel mill workers are occasionally assigned to spend a day repairing or changing the baghouse filters. Such a worker typically spends four to six hours in the midst of the suspended filters in the dust-laden atmosphere of the baghouse,[5] wearing a respirator equipped with a half-mask face piece. At a typical facility, this procedure is carried out an average of seven times per year. The analysis assumes that the same worker is assigned to this task every time. While performing such maintenance, the worker may be exposed to external radiation from the residual dust that is retained in the filters after they are emptied, as well as to the dust that has settled on the floor of the baghouse.

In addition, one worker typically spends about one hour per day monitoring the control mechanisms and performing maintenance that does not require entering the modules containing the filters. It is conservatively assumed that the same worker who maintains the filters would be assigned to this duty on days he were not inside the modules. The rest of the time, he would be assigned a variety of tasks in the steel mill. His external exposure rate during that time is assumed to be the same as that of the crane operator.[6] His internal exposure rate is assumed to be the average of workers inside a typical mill (see Appendix H).

The driver of the tanker truck transporting the dust off site may be exposed to external radiation from the dust in the truck. At one EAF facility which was visited, the dust is shipped to a processing facility about 60 miles away. Assuming an average speed of 40 miles per hour, one trip takes 1½ hours. Since the reference facility operating at full capacity produces about 2,250 tons of dust per year, a truck carrying 25 tons of dust would make 90 trips per year. Consequently, the driver would spend an average of 135 hours per year transporting this dust.

Slag Disposal

After the completion of the melt cycle, the EAF is tilted and the slag is poured into a ladle, which is moved by overhead crane to a slag yard outside the building. A worker at a typical facility spends about half his time on a platform on the edge of the slag yard and may be exposed to

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[5] Rest periods necessitated by work in a confined area and the need to don and remove protective clothing restrict the amount of time the worker can spend on this task.

[6] This worker was selected as having the median exposure rate to Co-60, one of the significant radionuclides in the present analysis.

external radiation from the slag. Since the rest of his time is in the vicinity of the slag, he would be exposed to slag dust during the course of the day.

Since the slag pile is exposed to the elements, soluble components of the slag leach out of the matrix and percolate through the soil until they reach an underlying aquifer. (This process takes a number of years—see Section 6.4.1.) A nearby resident who gets his drinking water from a well that is down-gradient from the slag pile may, at some time in the future, be exposed to contaminated ground water.

5.2.4 Use of Steel Mill Products

All products of the steel mill have industrial uses. The present analysis deals with two of these products: finished steel and slag. In addition, most baghouse dust is reprocessed to recover zinc and other valuable metals.

Slag

As shown in Appendix I, slag is primarily used in road building, as fill, or for soil conditioning. A worker employed in road construction may be exposed to external radiation from the slag in the roadbed as well as that in the cement pavement—he may also be exposed to contaminated slag dust. It is assumed that the fraction of the slag generated by the melting of residually contaminated scrap is the same as the fraction of contaminated scrap in the total scrap melted by the mill during the peak year (i.e., the dilution factor of slag is 9.1, the same as that of the steel).

Steel

Steel is used to make a virtually endless variety of finished products. The analysis considers the four categories of products which are listed below, along with a representative component or example of each category. These products also represent small, medium and large objects, as indicated below.

- Large home appliance (medium-sized object): double oven
- Automotive components (medium-sized objects): engine block, body shell [7]
- Large industrial equipment (large object): 8-ton metal-working lathe
- Cooking utensil (small object): frying pan

Only the the oven and auto body are made from carbon steel, however. The other three are made primarily of cast iron, which is produced by a different process. The radiation exposures of workers producing and assembling two of these products—engine blocks and industrial lathes—are assessed in the present analysis. In each case, the workers may be exposed to external radiation from the iron, which is assumed to have the same concentration factor as the steel. The grinding operations on the lathe bed may also expose the lathe maker to the inhalation and ingestion of iron dust.

End Users of Finished Products

The final group of exposed individuals is composed of people who use the products listed in the previous section. One representative user of each of the four products is included in the analysis. A consumer may be exposed to external radiation from the steel in the kitchen range. A taxicab driver may be exposed to external radiation from the shell of the auto body, while a lathe operator may be exposed to radiation from the cast iron lathe bed. Another consumer cooking food in a cast iron frying pan may be exposed to external radiation from the cast iron, in addition to eating food which may be contaminated with residual radioactivity that has leached from the pan.

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[7] The engine block was selected to be the representative automotive component for the manufacturing process because it comprises a large portion of the total mass and because an assembly worker spends much of his work day in close proximity to this source. Because of its large area, the sheet metal in the auto bodies has a greater potential for external exposure of the occupant of the automobile, and was therefore selected as the representative automotive component for the end user exposure scenario.

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CHAPTER 6

CALCULATION OF RADIOLOGICAL IMPACTS ON INDIVIDUALS

Chapter 5 presented the scenarios and modeling parameters used to assess the radiation exposures of individuals that may result from recycling steel scrap from nuclear facilities. Chapter 6 discusses how these scenarios, as well as the effluent gaseous emissions from the facility, are used to perform a radiological assessment of these individuals. For the sake of clarity in the presentation, the scrap dilution factor presented in Section 5.2 is not discussed in the present chapter, which assumes that all recycled scrap is potentially contaminated. The scrap dilution factor will be explicitly addressed in the discussion of results in Chapter 7.

The concept of the RMEI is central to the assessment—it is discussed here in more detail. For a single exposure scenario and a given radionuclide, such as scrap contaminated with a strong γ-emitting nuclide (e.g., cobalt-60 or cesium-137), the choice of the RMEI is relatively straightforward: it is the individual worker who spends the most time nearest to the scrap. For the entire life cycle of a given batch of scrap metal—from the time it leaves the custody of a licensed facility, is transported to a steel mill, is turned into sheet metal, is used to fabricate a kitchen range and finally is delivered to a home—there may be several exposed individuals. The RMEI is not obvious *a priori*. To determine which individual receives the highest exposure, the annual doses to the exposed individuals at each stage of production, transportation, distribution and storage, including the use of the finished product, are compared. The person with the highest dose rate would become the RMEI for a given radionuclide.

A number of computer codes dealing with recycling and pathways analysis were reviewed for use in this study but none were found entirely suitable. Initially, a series of computer spreadsheets were developed to perform the calculations described in this chapter. As the analysis progressed, the need for a single integrated computer program became evident. Such a program was therefore developed for this analysis. The program is written in the FORTRAN 90 computer language and can run on an IBM-compatible personal computer.

6.1 RADIOACTIVE CONTAMINANTS

The 40 individual radionuclides studied in this analysis were selected on the basis of a review of nine published reports which cast light on the nuclides most likely to be present in scrap metal that may be a candidate for recycling. A detailed discussion of the selection process is presented in Appendix D.

Since a period of years is assumed to elapse between the time the metal was contaminated and the time it would be recycled, short-lived nuclides (i.e. those with half-lives of less than 6 months) would have decayed to insignificant levels and were therefore omitted from the present analysis. By the same token, short-lived progeny of long-lived parents are assumed to be in secular equilibrium with their parents at the time of recycling. All references to such parent nuclides in this report include the designation "+D" to indicate that the contributions of this implicit progeny are included in the calculated annual doses and risks, which are normalized to unit specific activity *of the parent.* The implicit progenies of all nuclides selected for the present analysis are listed in Table 6-1. The generation number indicates whether the progeny nuclide is first generation (1), second generation (2), etc.

The analysis also considered steel scrap potentially contaminated with unique combinations of radionuclides, including long-lived members of natural decay series in secular equilibrium with their parents. These include: 1) "U-separated"—the three uranium isotopes (in secular equilibrium with their short-lived progenies but separated from their long-lived progenies) in the ratios of their natural abundances; 2) "U-depleted"—the same isotopes in ratios characteristic of depleted uranium;[1] 3) "U-natural"—natural uranium in secular equilibrium with the entire U-238 and U-235 radioactive decay series, and 4) "Th-series"—Th-232 in secular equilibrium with its entire decay series. The calculated radiological impacts of the mixtures of uranium isotopes, as well as those of the uranium series, are normalized to unit activities of U-238, while those of the thorium series are normalized to unit activities of Th-232. The nuclides included in each of these groupings are listed in Table 6-2.

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[1] Depleted uranium is a byproduct of the uranium enrichment process and contains reduced activities of U-234 and U-235.

Table 6-1. Implicit Progenies of Nuclides Selected for Analysis[a]

Parent			Progeny				
Nuclide	Half-Life	Radiation	Generation	Nuclide	Branching Ratio	Half-Life	Radiation
Sr-90	28.6 y	β	1	Y-90	100%	64.1 h	β
Ru-106	1.01 y	β	1	Rh-106	100%	29.9 s	β,γ
Ag-110m	249.8 d	β,γ,e^-	1	Ag-110	1.4%	24.6 s	β,γ
Sb-125	2.77 y	β,γ,e^-	1	Te-125m	22.8%	58 d	γ,e^-
Cs-137	30.2 y	β	1	Ba-137m	94.6%	2.52 m	γ,e^-
Ce-144	284 d	β,γ,e^-	1	Pr-144m	1.43%	7.2 m	γ,e^-
			1	Pr-144	98.6%	17.3 m	β,γ
Pb-210	22.3 y	β,γ,e^-	1	Bi-210	100%	5.01 d	β
			2	Po-210	100%	138 d	α
Ra-226	1600 y	α,γ,e^-	1	Rn-222	100%	3.82 d	α,γ
			2	Po-218	100%	3.05 m	α
			3	Pb-214	99.98%	26.8 m	β,γ,e^-
			4	Bi-214	100%	19.9 m	β,γ,e^-
			5	Po-214	99.97%	164 μs	α,γ
Ra-228	5.75 y	β	1	Ac-228	100%	6.13 h	β,γ,e^-
Ac-227	21.8 y	α,β,γ,e^-	1	Fr-223	1.38%	21.8 m	β,γ,e^-
			1	Th-227	98.6%	18.7 d	α,γ,e^-
			2	Ra-223	100%	11.4 d	α,γ,e^-
			3	Rn-219	100%	3.96 d	α,γ,e^-
			4	Po-215	100%	1.78 ms	α,γ
			5	Pb-211	100%	36.1 m	β,γ,e^-
			6	Bi-211	100%	2.13 m	α,γ,e^-
			7	Tl-207	99.7%	4.77 m	β,γ
			7	Po-211	0.27%	0.52 s	α,γ
Th-228	1.91 y	α,γ,e^-	1	Ra-224	100%	3.62 d	α,γ,e^-
			2	Rn-220	100%	55.6 s	α,γ
			3	Po-216	100%	0.15 s	α
			4	Pb-212	100%	10.6 h	β,γ,e^-
			5	Bi-212	100%	60.6 m	α,β,γ,e^-
			6	Tl-208	35.9%	3.05 m	β,γ,e^-
			6	Po-212	64.1%	0.30 μs	α

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[a] Only progenies with half-lives of six months or less are included in the implicit progeny of "+D" nuclides.

Table 6-1 (continued)

Parent			Progeny				
Nuclide	Half-Life	Radiation	Generation	Nuclide	Branching Ratio	Half-Life	Radiation
Th-229	7340 y	α,γ,e^-	1	Ra-225	100%	14.8 d	β,γ,e^-
			2	Ac-225	100%	10.0 d	α,γ,e^-
			3	Fr-221	100%	4.8 m	α,γ,e^-
			4	At-217	100%	32.3 ms	α,γ
			5	Bi-213	100%	45.7 m	α,β,γ,e^-
			6	Tl-209	2.16%	2.20 m	β,γ,e^-
			6	Po-213	97.8%	4.20 μs	α
			7	Pb-209	100%	3.25 h	β
U-235	7.06e+08 y	α,γ,e^-	1	Th-231	100%	25.5 h	β,γ,e^-
U-238	4.47e+09 y	α,e^-	1	Th-234	100%	24.1 d	β,γ,e^-
			2	Pa-234m	100%	1.17 m	β,γ,e^-
			3	Pa-234	0.16%	6.70 h	β,γ,e^-
Np-237	2.14e+06 y	α,γ,e^-	1	Pa-233	100%	27.0 d	β,γ,e^-
Pu-241	14.4 y	β	1	Am-241	100%	432 y	α,γ,e^-

Because of the variabilities of contamination patterns and storage conditions, it cannot be assumed that radon isotopes would escape from the surface of the metal. The contamination might have been painted over, for instance, or trapped inside a steel component that was crushed as part of a volume reduction process. Therefore, for example, the assessment of Ra-226+D assumes that Rn-222 and its short-lived daughter products would remain in the scrap in complete secular equilibrium with the radium, while that of natural uranium series assumes that both Rn-222 and Rn-219, as well as their entire progenies, would be in secular equilibrium with U-238 and U-235, respectively. Similarly, the assessments of the Th-232 series and of Th-228+D assume that Rn-220 would remain in the scrap.

Except for the "+D" nuclides with their short-lived progeny and the natural uranium and thorium series, no ingrowth of progenies was modeled in the radiological assessment of individuals. Exposures of scrap processors, steel mill workers and slag users are assumed to occur within a few months of the release of the scrap for recycling, too short a time for any significant ingrowth of long-lived progeny. Since the finished products that were among the subjects of the analysis have useful lives of several years, such ingrowth could potentially occur. However, as will be seen in the discussion of such exposure scenarios later in this chapter, such ingrowth would have no significant impact for the nuclides and the materials considered in the analysis.

Table 6-2. Nuclides Included in Various Combinations and Decay Series

Series	Nuclide	Activity Fraction
U-Natural	U-238+D	1.
	U-234	1.
	Th-230	1.
	Ra-226+D	1.
	Pb-210+D	1.
	U-235+D	0.047
	Pa-231	0.047
	Ac-227+D	0.047
U-Separated	U-238+D	1.
	U-234	1.
	U-235+D	0.047
U-Depleted	U-238+D	1.
	U-234	0.09123
	U-235+D	0.01594
Th-Series	Th-232	1.
	Ra-228+D	1.
	Th-228+D	1.

6.2 SPECIFIC ACTIVITIES OF VARIOUS MATERIALS

When steel scrap is charged to an electric arc furnace (EAF), chemical agents (fluxes) are added to control the chemical properties of the molten metal. The interactions among the flux, the refractory brick which lines the furnace, and the molten metal affect the final composition of the melt and hence the distribution of radionuclides among the several furnace products, such as the melt, the slag and the off-gas. The melt is subsequently allowed to cool and becomes the primary output of the mill. Slag is the material not remaining in the metal and includes the chemical agents, some of the liner material, and small amounts of the base metal, much of which is recovered and charged to the furnace for a subsequent melt. Off-gas consists of the fumes and aerosols evolved during melting which are captured by the facility's emission control system and, after cooling, collected in the baghouse in the form of dust.

To perform an exposure assessment of a given contaminant in the scrap, it is necessary to determine how that material is distributed in the various media products following the melting of the scrap.

The concentration of radionuclide *i* in medium *m* is calculated as follows:

$$C_{im} = \frac{C_{is} M_s P_{im}}{M_m} \tag{6-1}$$

C_{im} = specific activity of radionuclide *i* in medium *m* (pCi/g)
C_{is} = specific activity of radionuclide *i* in the scrap (pCi/g)
M_s = mass of radioactive scrap in furnace charge (g)
P_{im} = partition ratio or vaporization fraction of radionuclide *i* in medium *m*
M_m = mass of medium *m* produced from that charge (g)

A literature search as well as thermodynamic calculations were used to develop the partition ratios and vaporization fractions for EAF melting of carbon steel used in the present analysis which are listed in Table 6-3. Ranges of partition ratios reflect variability in melting practices. A detailed report of this study appears in Appendix E. A similar study was performed for cast iron production; it is reported in Appendix F.

To calculate the concentration factor, CF (i.e., C_{im}/C_{is}), it is also necessary to determine the mass of each medium as a fraction of the mass of the furnace charge. Based largely on the discussion in Section E.7 of Appendix E and a comparable discussion in Appendix F, the following mass fraction values were adopted for the present analysis:

- Imported scrap (scrap from sources outside the mill): 0.95
- Home scrap (metal recovered from by-products of previous melts): 0.05
- Finished steel: 0.9
- Steel slag: 0.117
- Cast iron slag: 0.065
- Baghouse dust: 0.015
- Melt: 0.97 (see below)

Table 6-3. Partition Ratios (PR) and Concentration Factors (CF)[a]

Element	Imported Scrap	Furnace Charge[b]	Steel		Cast Iron CF	Slag		Baghouse		Volatile PR (%)
			PR (%)	CF		PR (%)	CF	PR (%)	CF	
Ac	1	0.95	0	0	0	95	7.79	5	2.6	
Ag	1	1	99/75	1.02	1.01		0	1/25	16.5	
Am	1	0.95		0	0	95	7.79	5	2.6	
C	1	1	100/27	1.03	1.01		0		0	0/73
Ce	1	0.95		0	0	95	7.79	5	2.6	
Cm	1	0.95		0	0	95	7.79	5	2.6	
Co	1	1	99	1.02	1.01		0	1	0.67	
Cs	1	0.95		0	0	0/5	0.41	100/95	63.3	
Eu	1	0.95		0	0	95	7.79	5	2.6	
Fe	1	1	97	1	1	2	0.19	1	0.67	
I	1	0.95		0	0		0		0	100
Mn	1	0.98	24/65	0.67	0.98	72/32	6.15	4/3	2.21	
Mo	1	1	99	1.02	1.01		0	1	0.67	
Nb	1	0.95		0	0	95	7.79	5	2.6	
Ni	1	1	99	1.02	1.01		0	1	0.67	
Np	1	0.95		0	0	95	7.79	5	2.6	
Pa	1	0.95		0	0	95	7.79	5	2.6	
Pb	1	0.95		0	0		0	100	63.3	
Pm	1	0.95		0	0	95	7.79	5	2.6	
Po	1	0.95		0	0		0	100	63.3	
Pu	1	0.95		0	0	95	7.79	5	2.6	
Ra	1	0.95		0	0	95	7.79	5	2.6	
Ru	1	1	99	1.02	1.01		0	1	0.67	
Sb	1	1	99/80	1.02	1.01		0	1/20	13.2	
Sr	1	0.95		0	0	95	7.79	5	2.6	
Tc	1	1	99	1.02	1.01		0	1	0.67	
Th	1	0.95		0	0	95	7.79	5	2.6	
U	1	0.95		0	0	95	7.79	5	2.6	
Zn	1	0.96	20/0	0.2	0.02		0	80/100	63.3	

[a] Data is relevant to EAF operations, except cast iron concentration factors, which apply to iron foundries.

[b] Refers to the scrap metal charged to the furnace, which consists of 95% imported scrap and 5% recirculating home scrap.

The iron content of slag can be calculated from the average chemical composition of steel slags, which is listed in Appendix I. Comparing these data with the composition of baghouse dust, presented in Appendix E-2, allows us to infer the source of the dust. We thus conclude that:

- Fe content of slag = 19.4 %
- Source of baghouse dust: ⅓ slag, ⅔ steel

In deriving the concentration factors, the home scrap was assumed to have the same concentration as the melt. In all cases where the partition ratio (PR) for slag was listed as 95, it is assumed that all the activity first partitions to the slag, which initially has a mass fraction of 0.122, of which 0.005 forms the baghouse dust. The above calculations assumed that all dust is retained by the baghouse filters—the release of filtered particulates to the atmospheres are not included in the volatilization factors.

In all cases where a range of partition ratios is listed for a given chemical element in a given medium, the high end of the range is used to calculate the corresponding concentration factor. The range results from the variability of melting practices and other factors. Consequently, a given individual may be exposed to radionuclide concentrations corresponding to the high end of the range in one medium, while a different individual could be exposed to the high end of the range for a different medium. In only one scenario in the present analysis—the operator of the continuous caster—is the same individual exposed to radioactivity from two different media (other than scrap). As shown in Table 5-1, this individual would be exposed to external radiation from the steel while inhaling and ingesting the furnace emissions (i.e., baghouse dust). The radiological impacts on this individual of those nuclides that have a range of partition ratios in both the steel and the dust—isotopes of silver, manganese, antimony and zinc—are overstated. (Rather than calculating the partition ratios that would result in the maximum exposures in these few cases, these conservative estimates were retained in the analysis.) In all other cases, however, this approach yields a reasonable, maximum exposure assessment.

6.3 EXPOSURE PATHWAYS

6.3.1 External Exposures to Direct Penetrating Radiation

Table 5-1 shows that the external exposure pathway is included in every scenario except the consumption of ground water contaminated by leachate from a slag pile. Except for the assessment of exposure to airborne effluent emissions, which are discussed later in this chapter, external exposures were evaluated either by using the MicroShield™ computer code or by employing the external exposure dose coefficients in Federal Guidance Report (FGR) No. 12 (Eckerman 93).

Use of MicroShield™ Computer Code

MicroShield™ (Grove 95) is an industry-standard computer program used to perform γ-ray shielding calculations for radioactive sources. Results obtained with MicroShield™ are generally in good agreement with those performed by photon transport codes employing discrete ordinate or Monte Carlo methods. At photon energies below about 100 keV, the MicroShield™ results begin to diverge from those calculated by the more elaborate methods. This limitation, however, is not of concern in the present analysis. The primary dose contribution from most of the γ-emitting nuclides is from the high-energy photons. From nuclides that emit only low-energy photons, the dose is dominated by internal exposure. In neither case do the low-energy photons make a significant contribution to the total dose.

MicroShield™ utilizes dose coefficients listed in ICRP Publication 51 (ICRP 87) to calculate the effective dose equivalent for each of five exposure geometries. For most exposure scenarios, the present analysis assumes that the radiation is incident in the anterior-posterior direction, which corresponds to the exposed individual's facing the radiation source. This is a realistic assumption in most cases—it also results in the highest dose. The resulting dose conversion factors (DCFs), expressed in millirem per hour, are utilized in the assessments of the external exposure pathways. An illustrated description of the source and receptor configuration for each scenario analyzed with Microshield™ are presented in Appendix H.

The DCFs are used to calculate the normalized doses and risks from external exposure. The source-to-receptor distance and the duration of exposure for each scenario are listed in Table 5-1. Additional details are presented in Appendix H. The annual dose to the maximally exposed individual from a given nuclide in a given scenario is calculated by multiplying the appropriate DCF by the exposure duration and by the specific activity in the source medium, normalized to a unit specific activity in the scrap. The concentration factors for the various nuclides in the different media are listed in Table 6-3. These calculations are shown in Equation 6-2, below.

$$D_{tmx}(x) = C_{tm} F_{tmx}(x) \, t_e \qquad (6\text{-}2)$$

$D_{imx}(x)$ = dose from one year of external exposure to radionuclide *I* in medium *m* at distance *x* (mrem/y per pCi/g in scrap)

x = distance from source to receptor (m)

t_e = annual exposure duration (hr/y)

$F_{imx}(x)$ = DCF from external exposure at distance *x* from radionuclide *I* in medium *m* in a given source configuration (mrem·g/pCi·hr)

External Exposure over a Varying Distance

In several scenarios, such as the EAF furnace operator, the distance between the source of the external radiation and the exposed individual varies over time—i.e., the operator is at different locations during the course of the day. Although the minimum and maximum distances of a given individual have been observed or can be inferred, the duration of his or her occupancy of the various locations within this range is difficult to ascertain. The analysis therefore makes the simplifying assumption that the individual spends an equal amount of time at each distance. This is equivalent to assuming that he or she moves uniformly back and forth, like a sentry walking his post between two points.

To determine the integrated exposure during this time, it is necessary to derive the exposure rate at some arbitrary distance from the source, given the exposure rates at two fixed distances. To do this, we first calculate the distance and strength of a fictitious equivalent point source that would produce the same exposure rates at the same locations as those calculated for the real source. Applying the inverse square law, we obtain:

$$R(x) = \frac{A_o}{(x - x_o)^2} \qquad \text{(6-3)}$$

R(x) = exposure rate at distance *x* from real source

A_o = strength of equivalent point source

x_o = distance of equivalent point source from real source

To evaluate the constants A_o and x_o, we substitute the calculated values of R(x) at two known distances:

$$R_1 = \frac{A_o}{(x_1 - x_o)^2}$$

$$R_2 = \frac{A_o}{(x_2 - x_o)^2} \qquad (6\text{-}4)$$

Solving Equations 6-4, we obtain:

$$A_o = R_1 (x_1 - x_o)^2$$

$$x_o = \frac{x_1 R_1^{1/2} - x_2 R_2^{1/2}}{R_1^{1/2} - R_2^{1/2}} \qquad (6\text{-}5)$$

Next, we find the mean value of R(x) over the interval, $x_3 < x < x_4$

$$\overline{R} = \frac{A_o \int_{x_3}^{x_4} \frac{dx}{(x - x_o)^2}}{x_4 - x_3} \qquad (6\text{-}6)$$

$$= \frac{A_o}{(x_3 - x_o)(x_4 - x_o)}$$

Equation (6-6) is used to evaluate the factor $F_{imx}(x)$ in Equation 6-2 over the range $[x_3, x_4]$.

Use of FGR 12 Dose Coefficients

MicroShield™ is a useful tool for determining dose rates from relatively compact sources. In some scenarios, however, the external radiation comes from a planar source whose lateral dimensions are large in comparison to the source-to-receptor distance, and which has a mass thickness many times greater than the mean free path of the most penetrating radiation of any of the nuclides in the analysis. In those cases, the dose coefficients for soil contaminated to an infinite thickness listed in FGR 12 provide a convenient method of analysis.

These factors were applied to the slag yard worker standing at the edge of the slag. Since he is only exposed to one-half of an infinite plane, he would only get half the dose predicted by FGR 12. Since the average atomic number of slag is somewhat higher than that of soil, the analysis would tend to overstate the doses. For the nuclides with the most energetic γ-rays, for which external exposure is a major pathway, the interaction of the radiation with the source material is primarily by Compton scattering, which is relatively insensitive to the atomic number.

The FGR 12 dose coefficients were also used to evaluate the external exposure of the scrap cutter. Since he spends time in alleys surrounded by walls of scrap, it is reasonable to model the sources as two vertical half-planes beginning at the ground surface. The two half-planes together are equivalent to a single infinite plane. Again, the scrap has a higher atomic number than the average for soil, yielding a somewhat conservative but not excessively overstated assessment.

6.3.2 Inhalation of Contaminated Dust

During certain of the operations listed in Table 5-1, some of the radioactively contaminated material is assumed to be dispersed in the ambient atmosphere in the form of dust. The radiation exposure of an individual inhaling this dust will depend on his breathing rate, the dust loading of the ambient air, the respirable fraction (i.e., the mass fraction of particles with AMAD≤ 10 μm)[2], the exposure duration and on whether or not he uses some form of respiratory protection. The radiological impacts are modeled by the following equations:

$$\begin{aligned} D_{imh} &= B\ C_{im}\ f_f f_r\ F_{ih} t_e \chi_d \\ R_{imh} &= B\ C_{im}\ f_f f_r\ G_{ih} t_e \chi_d \end{aligned} \qquad (6\text{-}7)$$

D_{imh} = 70-year dose commitment from inhalation of radionuclide *I* in medium *m* during one year (mrem/y EDE per pCi/g in scrap)

B = breathing rate

= 1.2 (m^3/hr)

f_f = respiratory protection factor (filter factor, dimensionless)

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[2] AMAD is the acronym for Activity Median Aerodynamic Diameter, "[which] is the diameter of a unit density sphere with the same terminal settling velocity in air as that of an aerosol particle whose activity is the median of the entire aerosol." (Eckerman 88).

f_r = respirable fraction
F_{ih} = DCF for inhalation of radionuclide *I* (mrem/pCi—FGR 11)
χ_d = concentration of dust in air (dust loading, g/m^3).
R_{imh} = excess lifetime risk of radiogenic cancer from inhalation of radionuclide *I* in medium *m* during one year (y^{-1} per pCi/g in scrap)
G_{ih} = risk factor for inhalation of radionuclide *I* (pCi^{-1}—EPA 94a)

The dust loading for each exposure scenario is listed in Table 5-1. A discussion of the derivation of these values appears in Appendix H.

The analysis assumes that all of the airborne dust emanates from the contaminated material being processed and that the specific activity of a given radionuclide in the dust is the same as that of the material. This assumption is realistic for operations such as handling of baghouse dust or slag, or the use of a cutting torch on scrap. In these cases, the dust results from the operations and would contain the same radionuclides as the material in process.

Studies also show that the specific activity in the dust may be either greater or less than the radionuclide concentrations in the source of the dust due to enhancement and discrimination processes. For example, the particles that become airborne are usually less than 50 μm in diameter (Peterson 83). If most of the radioactivity in the material that is the source of the dust is found in particles larger than 50 μm, then activities in the dust are likely to be lower than that in the source of the dust. Conversely, if the activity is primarily on particles smaller than 50 μm, the specific activity in the dust can be greater than that in the source of the dust. The assumption that the specific activity of a given radionuclide in the dust is the same as that of the material is a reasonable approximation in most cases. Many other physical and chemical properties besides particle size can also produce enhancement or discrimination effects. A discussion of this subject is provided in Envirosphere 84.

The inhalation DCFs are based on Federal Guidance Report (FGR) No. 11 (Eckerman 88), while the inhalation risk factors (i.e., slope factors) are based on EPA 94a. These dose and risk factors are based on the assumption that the airborne particles have an AMAD of 1 μm, which results in the particles penetrating deeply into the lungs and creating the greatest potential for exposure. The larger particles that comprise the major part of the dust by mass do not penetrate as deeply into the lung and thus have a smaller radiological impact on the exposed individuals[3]. Accordingly, the dose and risk factors used in this study are conservative, upper-end values.

The dose factors also depend on the chemical form of the nuclide in question. Since the chemical form of the radionuclide on the scrap metal is unknown, it is assumed that a nuclide on or in the scrap has the form corresponding to the highest DCF listed in FGR 11. Similarly, it is assumed that the nuclides in the ground water will be in their most soluble form. In the case of baghouse dust, slag dust or vapors from molten metal, the analysis of radionuclide distributions during the melting of carbon steel indicates that in almost all cases the nuclides will be present as oxides. Consequently, the dose conversion factors corresponding to the respective lung clearance class and f_1 value, as listed in Table 3 of FGR 11, are adopted for the analysis. For the ingestion of radionuclides left in the melt—the frying pan scenario in the present analysis—the nuclides are assumed to be in the elemental form.

The chemical form of each element with radioactive isotopes that may be found in radioactively contaminated carbon steel is listed in Table 6-4, along with the appropriate lung clearance class and f_1 value.

6.3.3 Incidental Ingestion

Individuals working in a dusty, sooty environment are likely to inadvertently ingest some of the contaminated material, which is generically referred to in this report as soot. The radiological impacts of such incidental ingestion will depend on the soot ingestion rate of the exposed individual and on the duration of exposure. The impacts are modeled by the following equations:

$$\begin{aligned} D_{ing} &= C_{im} F_{ig} I_s \, t_e \\ R_{ing} &= C_{im} G_{ig} I_s \, t_e \end{aligned} \qquad (6\text{-}8)$$

---

[3] An additional contribution to the dose from dust inhalation is from large particles that are inhaled, refluxed from the air passages and then swallowed. In all scenarios where inhalation exposure is modeled, inadvertent ingestion is also assumed. Since the inadvertent ingestion rate of particulate material is typically several times larger than the inhalation rate of the same material, the ingestion of these large inhaled particles would not have a significant impact on the total dose.

D_{img} = 70-year dose commitment from ingestion of radionuclide *I* in medium *m* during one year (mrem/y EDE per pCi/g in scrap)
F_{ig} = DCF for ingestion of radionuclide *I* (mrem/pCi)
I_s = soot ingestion rate (g/hr)
R_{img} = excess lifetime risk of radiogenic cancer from ingestion of radionuclide *I* in medium *m* during one year (y^{-1} per pCi/g in scrap)
G_{ig} = risk factor for ingestion of radionuclide *I* (pCi^{-1})

The EPA Exposure Factors Handbook (EPA 89) presents a detailed discussion of soil and soot ingestion, primarily by children. However, data are also provided for inadvertent soil and soot ingestion rates by adults working in a dusty environment. For adults, the daily soil ingestion rates range from 0.56 mg/day for indoor work to 480 mg/d for outdoor work. Given the nature of the operations at scrap yards and steel mill, a mid-range value of 240 mg/d was assumed. Since the EPA values are assumed to refer to an 8-hour workday, a soot ingestion rate of 30 mg/hr was adopted for modeling the inadvertent ingestion pathway. The one exception is the lathe manufacturing operation, where it is likely that only part of the "dirt" in the area would come from the cast iron that is being ground. Some would come from the grinder itself, for instance. Consequently, an ingestion rate of 10 mg/hr was adopted for that operation.

Table 6-4. Lung Clearance Class and F_1 Values for Use with FGR 11

Element	Melt			Dust & Slag			Scrap & GW		
	Form	Class	f_1	Form	Class	f_1	Form	Class	f_1
Ac				Ac_2O_3	Y	E-3	max	D	E-3
Am				Am_2O_3	W	E-3	all	W	E-3
Ag	Ag	D	0.05	Ag	D	0.05	max	Y	0.05
C	C		1	CO_2		1	organic		1
Ce				Ce_2O_3	Y	3E-4	max	Y	3E-4
Cm				all	W	E-3	all	W	E-3
Co	Co	W	0.3	CoO	Y	0.05	max	W	0.3
Cs				all	D	1	all	D	1
Eu				all	W	E-3	all	W	E-3
Fe	Fe	W	0.1	FeO	W	0.1	max	D	0.1
I				all	D	1	all	D	1
Mn	Mn	D	0.1	MnO	W	0.1	max	W	0.1
Mo	Mo	D	0.8	MoO_3	Y	0.8	max	Y	0.8
Nb				Nb_2O_5	Y	0.01	max	Y	0.01
Ni	Ni	D	0.05	NiO	W	0.05	max	D	0.05

Table 6-4. Lung Clearance Class and F_1 Values for Use with FGR 11 (Continued)

Element	Melt			Dust & Slag			Scrap & GW		
	Form	Class	f_1	Form	Class	f_1	Form	Class	f_1
Np		W		all	W	E-3	all	W	E-3
Pa				PaO_2	Y	E-3	231 233/4	W Y	E-3
Pb				all	D	0.2	all	D	0.2
Pm				Pm_2O_3	Y	3E-4	max	Y	3E-4
Po				PoO_2	W	0.1	max	D	0.1
Pu				Pu_2O_3	Y	E-5	max	W	E-3
Ra				all	W	0.2	all	W	0.2
Ru	Ru	D	0.05	RuO_4	Y	0.05	max	Y	0.05
Sb	Sb	D	0.01	Sb_2O_3	W	0.01	max	W	0.01
Sr				SrO	D	0.3	[a]	D	0.3
Tc	Tc	W	0.8	TcO_2	W	0.8	max	W	0.8
Th				ThO_2	Y	2E-4	231/4 others	Y W	2E-4
U				UO_2	Y	2E-3	max	Y	0.05
Zn	all	Y	0.5	all	Y	0.5	all	Y	0.5

[a] All except $SrTiO_3$, an unlikely contaminant of potentially contaminated steel scrap

6.3.4 Radioactive Decay

Sections 6.3.1—6.3.3 present methods of calculating dose rates for all scenarios in which the source strength is essentially constant during the course of one year. These are the scenarios in which the source is replaced at frequent intervals. For scenarios in which the source is not replaced—the end user of finished products scenarios—radioactive decay over the course of a year must be taken into account. In such cases, the exposure is integrated over a period of one year, resulting in the following expression:

$$D_{imx}(x) = C_{im} F_{imx}(x) t_e \left(\frac{1 - e^{-\lambda_i t}}{\lambda_i} \right) \tag{6-9}$$

λ_i = radioactive decay constant of nuclide I (y^{-1})
t = integration time
= 1 y

The risk from external exposure is calculated by multiplying the corresponding dose by the risk factor for doses of low-LET radiation to the whole body:

$$R_{imx}(x) = D_{imx}(x)\, F_{Rx} \tag{6-10}$$

$R_{imx}(x)$ = excess lifetime risk of radiogenic cancer from one year of external exposure to radionuclide *I* in medium *m* at distance *x* (y^{-1} per pCi/g in scrap)

F_{Rx} = risk factor for external exposure

= $7.6\text{x}10^{-7}$ mrem^{-1} (EPA 94a)

In addition to decay, ingrowth of progeny was also considered. As stated earlier, all progenies with half-lives of less than six months are included in the exposure assessment of the parent. Eleven of the elements listed in Table 6-3 significantly partition to steel or iron. Mo-93 is the only one of isotopes of these elements that were included in the analysis that has a long-lived progeny, i.e., Nb-93m, which has a half-life of 16.1 y. The longest lived product is the industrial lathe, with a useful life of 20 years. Thus, even in the last year of anticipated use, the Nb-93m activity in the lathe would be less than 60% than that of Mo-93, which has a half-life of 3,500 y and would thus not decay significantly. The external dose from the Nb-93m approximately one-tenth of that from Mo-93, the inhalation dose would be about one-fourth and the ingestion about one-half. Given the other uncertainties in the analysis, omitting the Nb-93m contribution to the total Mo-93 dose in the finished product scenarios does not have a significant impact, especially since, as will be seen later, the individuals exposed to finished products potentially contaminated with Mo-93 receive a much smaller dose than the RMEI, selected from all scenarios in which Mo-93 is a potential contaminant.

6.4 UNIQUE SCENARIOS

The exposure assessment of two of the scenarios developed for the analysis required the development of special sub-models. These exposure pathways, the consumption of ground water contaminated by leachate from slag and the consumption of food cooked in a cast iron cookware made from potentially contaminated scrap, are described in the following sections, as is the impact of fugitive airborne emissions from the furnace on nearby residents. A qualitative discussion of the anticipated impacts of disposition of the recycled scrap in a sanitary landfill is presented at the end of Chapter 6.

6.4.1 Ground Water Contaminated by Leachate from Slag Storage Piles

As discussed in Section 5.2.2, an individual residing near the slag storage yard, who gets his drinking water from a well that is down-gradient from the slag could be exposed to contaminated ground water. During storage at the steel mill site, slag will be subjected to weathering and certain components may be leached from the slag and ultimately contaminate the local ground water. A study, based on a search of available literature, was performed to enable the calculation of leach rates of various radionuclides. Details of this study are presented in Appendix I. In addition, EPA is sponsoring an experimental study being conducted at the Brookhaven National Laboratory to determine the leach rates of constituents of various steel and iron slags. Some preliminary results of the Brookhaven study are presented in Appendix I-2. This section presents some of the information obtained from both studies and then develops an interim model of the leaching of radionuclides which partition to the slag.

Elemental Selection Criteria

Because of the scarcity of data, it was desirable to narrow the scope of the analysis to those elements that have radioactive isotopes which, if leached from slag produced by the melting of potentially contaminated steel scrap, would have a significant radiological impact via the ground water exposure pathway. The selection criteria include:

- potential contamination of steel scrap by one or more isotopes the given element
- significant partitioning of the given element to the slag (*i.e.*, concentration factor ≥ 0.1)
- travel time to the aquifer relative to half-life of longest-lived isotope being studied
- travel time relative to the 1,000 year period of impact assessment

Travel time. The travel time of the *I*-th nuclide was determined by Equation E.21 of the RESRAD user's manual (Yu 93).

$$\Delta t_i = \frac{\Delta z \; R_{d_i} \, p_e \, R_s}{I}$$

$$R_{d_i} = 1 + \frac{\rho_b \, K_{d_i}}{p_t \, R_s}$$

$$R_s = \left(\frac{I}{K_v} \right)^{\frac{1}{2b+3}}$$

Δz = thickness of unsaturated zone

= 4 m

R_{d_i} = retardation factor for *I*-th nuclide

p_e = effective porosity of unsaturated zone

= 0.2

I = infiltration rate

= 0.5 m/y

ρ_b = bulk soil density

= 1.5 g/cm^3

K_{d_i} = distribution coefficient for *I*-th nuclide (cm^3/g)

p_t = total porosity

= 0.485

K_v = saturated hydraulic conductivity of vadose zone

= 227 m/y

b = soil-specific exponential parameter

= 5.3

The values listed above, as well as the K_d's for soil, are those used for the generic site analysis in the TSD for the soil cleanup levels (EPA 94b).

Table 6-5. Potential Contaminants of Groundwater

Element	Slag CF	$t_{½}$[a] (y)	K_d	Δt_i (y)	Potential	Comments
Ac	7.79	21.8	240	1188.30	×	will decay
Am	7.79	432.7	1900	9400.32	×	will decay
Ce	7.79	0.8	500	2474.52	×	will decay
Cm	7.79	18.11	4000	19789.02	×	will decay
Cs	0.41	30.17	270	1336.71	×	will decay
Eu	7.79	13.6	240	1188.30	×	will decay
Fe	0.19	2.7	170	842.01	×	will decay
Mn	6.15	0.9	50	248.37	×	will decay
Nb	7.79	20,300	110	545.19	✓	
Np	7.79	2.14E6	5	25.76	✓	
Pa	7.79	32,760	110	545.19	✓	
Pb	progeny	22.26	270	1336.71		
Pm	7.79	2.6	240	1188.30	×	will decay
Pu	7.79	24,131	550	2721.87	×	Δt_i>> 1,000 y
Ra	7.79	1600	500	2474.52	×	Δt_i>> 1,000 y
Sr	7.79	28.6	15	75.23	✓	
Th	7.79	1.4E10	3200	15831.42	×	Δt_i>> 1,000 y
U	7.79	4.47E9	15	75.23	✓	

[a] Half-life of longest lived isotope

Elements with travel times longer than 20 half-lives[4] of their longest-lived isotopes or with travel times much greater than 1,000 years are marked with an "×" in Table 6-5, indicating that they are eliminated from consideration—it is not likely that any significant radioactivity would reach the aquifer during the 1,000-year assessment period under any reasonable environmental conditions.

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[4] Twenty half-lives was selected as the cutoff criterion since that represents a decay to 10^{-6} of the initial activity.

Slag Cement Leaching Studies

The American Nuclear Society has developed and formalized detailed procedures for measuring the leachability of solidified low-level radioactive wastes (ANS 86). This procedure involves testing of controlled geometry specimens in demineralized water at 17.5°C to 27.5°C to determine releases over successive intervals of time. Mass transport is assumed to be controlled by a diffusion process. When the fraction leached from a uniform sample is less than 20% of the initial activity, the leaching behavior can be approximated by that of a semi-infinite medium where the "effective diffusivity" is given by the following equation:

$$D_i = \pi T_n \left(\frac{a_{in} V}{A_{io} \Delta_{nt} S} \right)^2 \tag{6-11}$$

D_i = effective diffusivity of nuclide *I* (cm^2/s)
T_n = mean time of the leaching interval *n* (d)
= $\left(\frac{t_n^{1/2} + t_{n-1}^{1/2}}{2} \right)^2$
a_{in} = activity of nuclide *I* released during time interval *n* (g)
V = sample volume (cm^3)
A_{io} = initial activity of nuclide *I* in sample
Δ_{nt} = duration of *n*-th leaching interval (d)
= $t_n - t_{n-1}$
S = surface area of sample (cm^2)

When the cumulative fraction leached, $\Sigma_n \frac{a_{in}}{A_{io}}$, is greater than 0.2, Equation 6-11 must be corrected for specimen geometry.

Using a model and procedures similar to those described in ANS 86, Japanese investigators have determined the fractional leaching of Sr-90, Co-60, Cs-137 and H-3 from cement/slag composites in deionized water and synthetic sea water (Matsuzuru 77, 77a, 79). The duration of the leaching tests was about 100 days. The radionuclides were incorporated into the cement via a sodium sulfate solution. The composition of the slag is listed in Table 6-6.

Leaching data were analyzed using a plane source diffusion model to derive the expression

$$f_i = \frac{2S}{V}\sqrt{\frac{D_i t}{\pi}} \tag{6-12}$$

f_i = fraction of nuclide *I* leached in *t* days.

Equation 6-12 can be rewritten as

$$\begin{aligned} f_i &= \left[\frac{2S}{V}\sqrt{\frac{D_i}{\pi}}\right] t^{1/2} \\ &= m_i t^{1/2} \end{aligned} \tag{6-13}$$

where we have represented the expression in the square brackets by m_i, the slope of the line obtained by plotting f_i *vs.* $t^{1/2}$. Once m_i is determined, Equations 6-13 can be solved for D_i:

$$D_i = \left[\pi\left(\frac{V}{2S}\right)^2\right] m_i^2 \tag{6-14}$$

Since the actual leaching process involves an initial rapid leaching rate of a few days' duration (~ 7 d for Sr-90 and ~ 2 d for Co-60), followed by a longer term linear relation between f_i and $t^{1/2}$,

the experimental data are fitted to an equation of the form

$$f_i = m_i t^{1/2} + \alpha_i \tag{6-15}$$

Because of certain limitations and problems such as the initial leach rate, Matsuzuru *et al.* defined L, the leaching coefficient, with the same mathematical form as D in Equation 6-14.

Table 6-6. Composition of Slag Used in Leaching Test

Component	Composition (wt %)
SiO_2	28.7
Al_2O_3	11.5
Fe_2O_3	2.3
CaO	50.9
MgO	3.2
Insoluble Residue	0.8
Ignition Loss	0.6

Equation 6-12 can also be used to determine the values of f_i for various geometries, as follows:

$$f_{i2} = f_{i1}\left(\frac{S_2 V_1}{S_1 V_2}\right) \qquad (6\text{-}16)$$

where subscript 1 refers to geometry 1 while subscript 2 refers to geometry 2.

Values of L for Sr-90 leached from slag cements ranged from 1.2×10^{-7} to 1.7×10^{-7} cm^2/day for both deionized water and synthetic sea water at 25°C (Matsuzuru 77a). Using average values of L_{Sr} for samples cured seven days prior to testing in deionized water, and assuming a right circular cylinder, D = h, V = 70 cm^3 (r = 2.233 cm), we have derived values for m_{Sr} and α_{Sr} in Equation 6-15, which are listed in Table 6-7. The leachability of Cs-137 was reported to be about ten times that of Sr-90; therefore we assumed that $m_{Cs} = 10\ m_{Sr}$ and $\alpha_{Cs} = 10\ \alpha_{Sr}$. We then used Equation 6-16 to derive values that describe leaching from slag particles that are also right circular cylinders, but only 1 cm in diameter—a more representative size for EAF slags.

Table 6-7. Leaching Parameters Values

Element	m ($d^{-1/2}$)		α	
	r = 2.233 cm	r = 0.5 cm	r = 2.233 cm	r = 0.5 cm
Sr	5.8E-4	2.59E-3	4.97E-3	2.22E-2
Cs	5.8E-3	2.59E-2	4.97E-2	0.222
Cr[a]	7E-6	3E-5	0	0

[a] Cr is used as a surrogate for Nb, Np, Pa and U—see discussion below.

Use of the data obtained from slag cement leaching studies is believed to be conservative since the radionuclides in the cement composites are not dissolved in the slag and therefore not expected to be as tightly bound in the solid matrix. The strontium data were replaced by the data obtained from the Brookhaven National Laboratory, as described below.

Preliminary Data from Brookhaven National Laboratory

Preliminary results from leaching experiments on EAF slags performed at the Brookhaven National Laboratory indicate that the leaching of strontium, the only element checked in Table 6-5 that was measured in the leachate, was governed by diffusion (Fuhrmann 97). The diffusion coefficients determined from tests on three monolithic samples of EAF slag are listed in Table 6-8. We calculate the value of m_{Sr} for a monolithic cylinder beginning with an inversion of Equation 6-14:

$$m_i = \frac{2S}{V}\sqrt{\frac{D_i}{\pi}} = \frac{12}{r}\sqrt{\frac{D_i}{\pi}}$$

$$S = 2\pi\left(r^2 + hr\right) = 6\pi r^2 \qquad (h = 2r)$$
$$V = \pi r^3$$

m_{Sr} = $1.99 \times 10^{-2}\ d^{-1/2}$

D_{Sr} = $2.5 \times 10^{-11}\ cm^2/s$

= $2.16 \times 10^{-6}\ cm^2/d$

r = 0.5 cm

Table 6-8. Diffusion Coefficients for EAF Slag Monolithic Samples[a]

Slag Sample	Diffusion Coefficient (cm^2/s)
AS-1	1.4E-11
AS-2	2.5E-11
AS-3	6.2E-12

[a] from Fuhrmann 97

Since Fuhrmann did not report any initial releases that were not diffusion-controlled, α_{Sr} is assumed to equal zero. These new data were used to model the Sr-90 leaching from the slag in the present analysis.

Other Slag Leaching Studies

This section describes earlier leaching studies done on pure slags rather than slag/cement composites.

Australian researchers at CSIRO incorporated the toxic elements As, Sb, Cd, Zn and Cr into various types of slags by melting at 1300°C, and subsequently leached the slags according to the EPA TCLP protocol (Jahanshahi 94). In the TCLP test, a sample of at least 100 g, which has a minimum surface area of 3.1 cm^2/g or passes through a 9.5 mm sieve, is treated with about 2,000 g of extractant for 18 ± 2 hr at 22 ± 3°C using rotary agitation. The extractant has a pH of either 4.93 or 2.88, depending on the basicity of the sample (40 CFR 261, Appendix II, Method 1311). The pH is achieved by use of acetic acid which is buffered with sodium acetate for the higher pH level (55 FR 11798).

Slag samples were prepared by both slow cooling and quenching. Examination of the slag samples with an optical microscope showed that interconnecting porosity was present in the slow cooled and most of the quenched samples. Slow-cooled slag samples were crushed to either a "coarse" size (100% minus 10 mm)[5] or a "fine" size (100% minus 1 mm) for the leaching tests. In generalizing on the results of the TCLP tests, the researchers observed that:

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[5] "100% minus 10 mm" means that 100% of the particles passed through a screen with a 10 mm mesh.

- As and Sb leached more readily than Cd, Cr or Zn
- Fine particles generally leached more readily than coarse particles
- Slow cooled samples showed similar behavior to quenched samples

Based on the information presented in Jahanshahi 94, we estimated the fraction leached using the following assumptions:

- Slag compositions from Table III of Jahanshahi 94
- Sample size = 100 g
- Extractant volume = 2 L

The results are presented in Table 6-9. For three of the slags (CaFe1, CaFeSi1, and FeSi1), the compositions are markedly dissimilar to those expected from EAF melting of carbon steel. The other three slags, while not identical to EAF slags, are useful for developing preliminary modeling parameters. Unfortunately, of the five elements studied, only Cr is expected to be found in the slag in any significant quantity. However, in the absence of element-specific leaching data, Cr can be considered as a surrogate for the stable oxides expected in slags. Assuming that the fraction leached is proportional to $t^{1/2}$, this fraction can be expressed by the second line of Equation 6-13, where the upper limit of m_{Cr} is about $7 \times 10^{-6}\ d^{-1/2}$, based on Cr in the BF2 slags and an 18-hr leach test.

Leach Rate

The leach rate from slag is modeled by Equation 6-15, using the parameter values for 1-cm diameter particles listed in Table 6-7.

The mill is assumed to produce 150,000 tons of steel per year and 17,600 metric tons of slag as a by-product. The slag is assumed to be continuously dumped onto a 1-meter high pile and to be removed at the same rate, but with a 1-year inventory always remaining in place. The new slag is mixed uniformly with the old slag—the slag that is removed is thus a representative sample of this mixture. To model the age-dependent leach rate, we must first determine the age distribution of the individual particles in the pile. If there is a constant number, *N*, of particles in the pile, the number of particles removed during time *dt* is

$$dn = \lambda_r N \, dt$$

λ_r = removal rate constant
= 1/365
= $2.74 \times 10^{-3}\ d^{-1}$

Table 6-9. Fraction of Various Toxic Elements Leached from Slags Using EPA TCLP Protocol

Slag	Fraction Leached				
	As	Sb	Cd	Cr	Zn
CaFe1	3.48E-03	4.21E-05	3.10E-04	0.00E+00	3.00E-05
CaFeSi1	3.53E-03	2.68E-04	2.40E-04	0.00E+00	2.70E-05
CaFeSi2	5.09E-04	2.37E-04	6.80E-05	5.63E-07	2.3E-05
FeSi1	1.54E-04	1.10E-04	1.15E-04	4.82E-07	2.30E-5
BF1	1.68E-04	1.03E-04	1.10E-04	0.00E+00	1.34E-04
BF2	9.80E-04	4.29E-04	1.20E-03	6.00E-06	1.23E-03

Assume that v_o particles are added to the pile at some initial time ($t = 0$). After time t, the number of these particles left in the pile is given by

$$v(t) = v_o e^{-\lambda_r t}$$

By definition, this is the number of particles older than t. The number of particles with ages between t and $t+dt$ is obtained by differentiating the above expression with respect to t and changing the sign,

$$dv = \lambda_r v_o e^{-\lambda_r t} \, dt$$

Since this expression is independent of the initial time, we can generalize it to all the particles in the pile:

$$\frac{dn}{N} = \lambda_r e^{-\lambda_r t} dt \qquad (6\text{-}18)$$

dn = number of particles in pile with ages between *t* and *t+dt*

N = total number of particles in pile

The time-dependent leach rate is derived by differentiating f_i in Equation 6-15 with respect to time:

$$l_i = \frac{df_i}{dt} = \frac{m_i t^{-1/2}}{2}$$

Multiplying the above expression by the age distribution function of Equation 6-17 and integrating over the entire distribution

$$\begin{aligned} L_i' &= \int_0^\infty \frac{m_i \lambda_r t^{-1/2} e^{-\lambda_r t}}{2} dt \\ &= \frac{m_i \sqrt{\pi \lambda_r}}{2} \end{aligned} \qquad (6\text{-}19)$$

L_i' = leach rate of nuclide *i* in slag pile

Since the resulting leach rate is time-independent, the fraction leached during one year is obtained by multiplying the above result by the leaching time and adding the constant term in Equation 6-15,

$$\begin{aligned} F_i &= \frac{m_i T \sqrt{\pi \lambda_r}}{2} + \alpha_i \\ &= \frac{m_i \sqrt{\pi T}}{2} + \alpha_i \qquad \left(\lambda_r = \frac{1}{T}\right) \end{aligned} \qquad (6\text{-}20)$$

F_i = fraction of nuclide *i* in slag leached in time T

T = 365 d

The concentration in the pore water percolating through the soil (prior to any radioactive decay) is given by:

$$C_{ip} = \frac{C_{ig}\, D\, F_i\, \rho_g}{I} \tag{6-21}$$

C_{ip} = concentration of nuclide *i* in pore water (pCi/mL)

C_{ig} = specific activity of nuclide *i* in slag

D = depth of slag layer

= 100 cm

ρ_g = specific gravity of slag

= 2

Dilution. Dilution is modeled by the first of Equations E.27 in the RESRAD manual (Yu 93):

$$f = \frac{I\, l}{d_w K_a i}$$

f = dilution factor[6] (concentration in groundwater ÷ concentration in pore water)

= 0.142

l = length parallel to aquifer flow

= $\sqrt{A}$

= 94 m

A = area of contaminated zone

= V/D

= 8,845 m^2

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[6] Also called the dilution attenuation factor (DAF)

V = slag volume
= M/ρ
= 8,845 m^3
M = one year's slag production
= 150,000 tons steel/y x 0.13 (wt slag/wt steel)
= 17,690 Mg slag
ρ = specific gravity of slag
= 2
D = depth of slag layer
= 1 m
d_w = screened depth of well
= 3 m
K_a = saturated hydraulic conductivity of aquifer
= 5500 m/y
I = hydraulic gradient
= 0.02

The values of the last three parameters were also taken from the generic site analysis in EPA 94b.

Exposure Assessment

To calculate the dose and risk to an individual drinking the contaminated water, we combine the expressions in Equations 6-19 and 6-20 with the ground water dilution factor shown above. The concentration of a given radionuclide in the ground water is the product of these three expressions multiplied by an expression for radioactive decay during the travel time to the aquifer, Δt_i, which is determined by Equation E.21 of the RESRAD user's manual (Yu 93). (The values of Δt_i of all elements with radioisotopes that are included in the present analysis are listed in Table 6-5.) The dose or risk to the RMEI is determined by multiplying this concentration by the appropriate dose or risk factor and the drinking water consumption rate,

$$D_{ig} = C_{ip} \, f \, F_{ig} \, I_w \, e^{-\lambda_i \Delta t_i}$$
$$R_{ig} = C_{ip} \, f \, G_{ig} \, I_w \, e^{-\lambda_i \Delta t_i} \tag{6-22}$$

I_w = annual consumption of water
= 7.3×10^5 mL/y

Buildup of radioactive progeny

The long travel time of some radionuclides necessitates the consideration of ingrowth of their long-lived radioactive progeny. Of the five elements listed in Table 6-5 which have radioisotopes capable of reaching the aquifer, only three—neptunium, protactinium and uranium—have isotopes which in turn have radioactive progeny with half-lives greater than six months.

Neptunium. Table 6-5 shows that it would take neptunium leached from the slag less than 26 years to reach the aquifer. No significant ingrowth of the long-lived progeny of Np-237, the only neptunium isotope included in the present analysis, would occur during that time.

Protactinium. Pa-231, the parent of Ac-227, which has a half-life of 22.8 years, is the only nuclide which would have significant ingrowth of long-lived daughter products during its travel time of 545 years. Actinium has a higher K_d than protactinium and would thus travel more slowly. Still, its short half-life in comparison with the travel time of the parent indicates that significant daughter product activity would be found in the aquifer along with the parent. An upper bound of the radiological impact of Ac-227 can be calculated by assuming that actinium has the same K_d as protactinium, so that the two nuclides would be in secular equilibrium in the aquifer.

Uranium. Table 6-5 shows that it would take uranium leached from the slag 75 years to reach the aquifer. No appreciable progeny ingrowth from any of the three uranium isotopes included in the steel scrap recycling analysis would occur during that time.

6.4.2 Ingestion of Food Prepared in Contaminated Cookware

One of the finished products examined in this study is cast iron cookware made from potentially contaminated scrap metal. Radioactivity may leach from such cookware into the food and subsequently be ingested. The metal content of food cooked in cast iron cookware can be inferred directly from data in Reilly 85. From the listed concentrations of iron in beef and cabbage cooked in cast iron utensils, one can derive a weighted average of 13.5 ± 4.7 mg/kg. The equations used to calculate the dose and risk for this exposure pathway are:

$$\begin{aligned} D_{ig} &= C_{im}\, C_{mf} I_f F_{ig} \\ R_{ig} &= C_{im}\, C_{mf} I_f G_{ig} \end{aligned} \qquad (6\text{-}23)$$

C_{mf} = concentration of iron in food
= 1.35×10^{-5} g/g
I_f = amount of food ingested annually
= $I_b + I_v$
= 1.45×10^{5} g/y
I_b = beef consumption rate
= 7.5×10^{4} g/y
I_v = vegetable consumption rate
= 7.0×10^{4} g/y

6.4.3 Impact of Fugitive Airborne Emissions from the Furnace on Nearby Residents

The impact of fugitive airborne emissions from the furnace on nearby residents was modeled by means of EPA's Clean Air Assessment Package, using the computer code CAP88-PC.

To calculate the effects of airborne effluent emissions on the RMEI, the map showing locations of EAF facilities and commercial nuclear power plants and shutdown dates was used to identify seven EAF facilities which could receive the scrap from two or more nuclear plants in a single year. The meteorological data accompanying CAP88-PC was surveyed to identify the meteorological station nearest to each of these seven facilities. CAP-88 analyses for releases of C-14 and I-129 were performed using each of the meteorological data sets—the highest individual doses from the seven runs were used in the analysis. The RMEI was assumed to reside 1 km from the emission point—default CAP-88 values were used for all other parameters. Summaries of the CAP-88 analyses can be found in Appendix H.

CAP-88 calculates the risk of cancer fatality, not the risk of cancer incidence, which is the risk calculated for all other pathways in the present analysis. Furthermore, CAP-88 does not employ EPA's current risk factors, which are presented in EPA 94a. The CAP-88 dose calculations were used to estimate the lifetime risk of cancer incidence from one year's exposure to furnace emissions as follows:

$$R_{ij} = \frac{D_{ij} G_{ij}}{F_{ij}} \qquad (6\text{-}24)$$

R_{ij} = lifetime risk of cancer incidence from one year's exposure to nuclide *i* via pathway *j*

D_{ij} = 50-year dose commitment from one year's exposure to nuclide *i* via pathway *j*, as calculated by CAP-88 (mrem)

G_{ij} = risk factor for exposure to nuclide *i* via pathway *j*, from EPA 94a

F_{ij} = dose conversion factor for exposure to nuclide *i* via pathway *j*, from FGR 11 and 12 (Eckerman 88, 93)

6.4.4 Potential Doses to Individuals Following Disposal of Recycled Metal

The present analysis reflects a broad range of operations and exposure scenarios that could result from the recycling of steel scrap from nuclear facilities. Not included among these are the exposures that may be associated with the final disposal of the recycled metal in a municipal landfill. A quantitative analysis of the potential doses from this stage in the life cycle of recycled metal is not considered necessary because the individual doses cannot exceed those associated with the actual handling of the material at the steel mill or the exposure of individual users of the end products fabricated from recycled metal. As indicated in the discussion of each of the operations, workers at steel mills and end users of products made from recycled material can come into close contact with the radionuclides potentially in recycled scrap for extended periods of time. Recycled metal, if disposed of in a sanitary landfill where it is virtually precluded from prolonged contact with people, would not likely result in radiation exposures of individuals that exceed those of the RMEI identified in the present analysis.

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CHAPTER 7

RESULTS AND DISCUSSION OF RADIOLOGICAL IMPACTS ON INDIVIDUALS

This chapter presents a summary of the potential impacts of recycling scrap from nuclear facilities on the RMEI, as well as a brief discussion of the results of these analyses. The dose and risk from each radionuclide and each pathway are tabulated for every exposure scenario in Appendix J. The same results are tabulated more concisely by exposure pathway in Appendix K. A semi-quantitative evaluation of the uncertainties in the results is found in Chapter 10, while a more detailed discussion is presented in Appendix L.

7.1 NORMALIZED DOSES AND RISKS TO THE RMEI

The annual doses and risks for each scenario and for each radionuclide, normalized to unit specific activities in the scrap, are presented in Appendix K. Table K-1 lists the normalized doses to the maximally exposed individual worker from the radionuclides likely to be found on potentially contaminated steel scrap, calculated for each scenario described in Chapter 5. (The mnemonic code for each operation that occurs in this and subsequent tables is identified in Table 5-1.) Tables K-2 to K-4 list the contributions to the dose for each exposure pathway: external radiation, inhalation and ingestion. The corresponding lifetime risks of cancer resulting from one year's exposure to the same operations are listed in Tables K-5 to K-8.

These tables can be used for two purposes. First, if the average specific activity of a given nuclide in scrap is known, the annual dose or lifetime risk of cancer from one year's exposure in a given scenario can be calculated by multiplying the specific activity by the appropriate normalized dose or risk. If several radionuclides are present, the doses or risks are determined by summing the contributions of each nuclide. Second, the tables identify the scenario leading to the maximum dose or risk from each radionuclide. Assume, for example, that the maximum permissible specific activity of Co-60 in steel scrap released for recycling that would not cause any member of the general population to receive a dose greater than 5 mrem EDE[1] is to be determined. Table K-1 shows that the maximum dose, normalized to a unit specific activity (1 pCi/g), is 0.899 millirem in one year, and that that dose is received by the operator of the industrial lathe. Thus, the maximum specific activity that would limit the dose of the lathe

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[1] This is a purely hypothetical limit for didactic purposes only. No dose limit values are assumed in the present analysis.

operator, who has been identified as the RMEI for Co-60, to 5 millirem in any one year is:

$$C_{Co-60}(D^{max}) = \frac{D^{max}}{D'_{Co-60}}$$

$C_{Co-60}(D^{max})$ = maximum specific activity of Co-60 corresponding to maximum permissible dose, D^{max}

= 5.56 pCi/g

D^{max} = 5 mrem EDE

D'_{Co-60} = maximum annual normalized dose from C0-60

= 0.899 mrem EDE per pCi/g

7.2 MAXIMUM EXPOSURE SCENARIOS

Table 7-1 lists the scenario which would result in the maximum annual dose to the RMEI in that scenario for each radionuclide in the present analysis, as well as the dose and potential risk to that individual. Several observations can be made about the data in Table 7-1:

- The maximum annual normalized dose varies by more than six orders of magnitude, from a low of 4×10^{-6} mrem EDE per pCi/g from Ni-59 to a high of 8 mrem from Ac-227+D. Ten of the 44 nuclides and nuclide combinations studied would produce maximum doses greater than 1 mrem, while 20 others would be in the range of 0.1 to 1 mrem.

- Workers are the RMEI for almost all nuclides. The three exceptions are:

Table 7-1. Maximum Exposure Scenarios and Normalized Impacts on the RMEI from One Year of Exposure

Nuclide	Maximum Scenario		Annual Dose (mrem EDE per pCi/g)	Lifetime Risk of Cancer (per pCi/g)
	Description	Mnemonic		
C-14	Airborne effluent emissions	AIRBORNE	8.66e-04	4.28e-10
Mn-54	Lathe operator	OP-LATHE	2.02e-01	1.54e-07
Fe-55	Cutting scrap	SCRAPCUT	6.69e-06	2.71e-12
Co-60	Lathe operator	OP-LATHE	8.99e-01	6.84e-07
Ni-59	Cutting scrap	SCRAPCUT	4.39e-06	1.55e-12
Ni-63	Cutting scrap	SCRAPCUT	1.07e-05	4.41e-12
Zn-65	Cutting scrap	SCRAPCUT	9.61e-02	7.31e-08
Sr-90+D	Slag leachate in groundwater	SLGLEACH	1.52e+00	5.51e-07
Nb-94	Slag pile worker	SLAGPILE	4.74e-01	3.60e-07
Mo-93	Cutting scrap	SCRAPCUT	5.65e-05	1.17e-11
Tc-99	Cutting scrap	SCRAPCUT	2.15e-05	1.41e-11
Ru-106+D	Lathe operator	OP-LATHE	5.16e-02	3.93e-08
Ag-110m+D	Lathe operator	OP-LATHE	6.29e-01	4.78e-07
Sb-125+D	Cutting scrap	SCRAPCUT	6.37e-02	4.85e-08
I-129	Airborne effluent emissions	AIRBORNE	7.91e-01	5.04e-07
Cs-134	Cutting scrap	SCRAPCUT	2.46e-01	1.87e-07
Cs-137+D	Cutting scrap	SCRAPCUT	8.91e-02	6.77e-08
Ce-144+D	Slag pile worker	SLAGPILE	1.77e-02	1.36e-08
Pm-147	Slag pile worker	SLAGPILE	1.42e-04	8.31e-11
Eu-152	Slag pile worker	SLAGPILE	3.44e-01	2.61e-07
Pb-210+D	EAF furnace operator	FURNACE	3.08e+00	4.37e-07
Ra-226+D	Slag pile worker	SLAGPILE	6.27e-01	4.36e-07
Ra-228+D	Slag pile worker	SLAGPILE	3.68e-01	2.36e-07
Ac-227+D	Cutting scrap	SCRAPCUT	8.00e+00	1.35e-07
Th-228+D	Slag pile worker	SLAGPILE	1.35e+00	6.17e-06
Th-229+D	Slag pile worker	SLAGPILE	4.37e+00	2.32e-07
Th-230	Slag pile worker	SLAGPILE	6.42e-01	3.44e-08
Th-232	Slag pile worker	SLAGPILE	2.84e+00	3.34e-08
Pa-231	Slag pile worker	SLAGPILE	2.51e+00	5.20e-08
U-234	Slag pile worker	SLAGPILE	3.14e-01	3.31e-08

Table 7-1. Maximum Exposure Scenarios and Normalized Impacts on the RMEI from One Year of Exposure (Continued)

Nuclide	Maximum Scenario		Annual Dose (mrem EDE per pCi/g)	Lifetime Risk of Cancer (per pCi/g)
	Description	Mnemonic		
U-235+D	Slag pile worker	SLAGPILE	3.28e-01	5.90e-08
U-238+D	Slag pile worker	SLAGPILE	2.89e-01	3.55e-08
Np-237+D	Slag pile worker	SLAGPILE	1.53e+00	1.36e-07
Pu-238	Slag pile worker	SLAGPILE	6.82e-01	4.78e-08
Pu-239	Slag pile worker	SLAGPILE	7.29e-01	4.73e-08
Pu-240	Slag pile worker	SLAGPILE	7.29e-01	4.73e-08
Pu-241+D	Slag pile worker	SLAGPILE	1.17e-02	4.01e-10
Pu-242	Slag pile worker	SLAGPILE	6.93e-01	4.46e-08
Am-241	Slag pile worker	SLAGPILE	1.21e+00	1.07e-07
Cm-244	Slag pile worker	SLAGPILE	6.75e-01	6.69e-08
U-Natural	EAF furnace operator	FURNACE	3.61e+00	4.78e-07
U-Separated	Slag pile worker	SLAGPILE	6.18e-01	7.14e-08
U-Depleted	Slag pile worker	SLAGPILE	3.22e-01	3.95e-08
Th-Series	Slag pile worker	SLAGPILE	4.55e+00	8.86e-07

- Sr-90 which, due to its high leachability and low K_d, will readily leach from the slag pile into the ground water. For this nuclide, the RMEI would be a person living near the slag storage yard whose drinking water comes from a potentially contaminated well.

- C-14, which is potentially volatile (as CO_2) and would thus be released to the atmosphere and be potentially incorporated in food crops and animal fodder grown in the vicinity of the steel mill. The RMEI would be a person who gets a large portion of his food from these crops and farm animals.

- I-129, which is volatile and would also be released to the atmosphere and potentially contaminate food crops and animal fodder grown in the vicinity of the steel mill. The RMEI would again be a person who gets a large portion of his food from these crops and farm animals.

- Six scenarios account for the maximum doses from all 44 nuclides and nuclide combinations. The ground water potentially contaminated by slag leachate and the airborne effluent emissions scenarios are discussed above, the remaining four scenarios are discussed in the following sections.

7.2.1 Slag Pile Worker

The slag pile worker would receive the highest doses from many radionuclides that concentrate in slag, including Nb-94, Ce-144+D, Pm-147, Eu-152, radium and all the actinides except Ac-227. For the strong γ-emitters—Nb-94, Ce-144+D, Eu-152 and the two radium isotopes—the primary pathway is external exposure. This results from the worker's spending four hours per day exposed to slag in the slag yard—a massive source in close proximity. For the remaining nuclides, the primary pathway is inhalation of slag dust.

7.2.2 Cutting Scrap

The worker cutting scrap at the scrap yard would receive the highest doses from many of the nuclides that do not concentrate in the slag: Fe-55, isotopes of nickel, Zn-65, Mo-93, Tc-99, Sb-125 and isotopes of cesium. For Zn-65, Sb-125 and cesium, which are strong γ-emitters, the primary pathway would be external exposure. For the other nuclides, the primary pathway is dust inhalation.

Like the slag pile worker, the scrap cutter would be in close proximity to massive quantities of potentially contaminated material. His use of a cutting torch causes the metal to volatilize, potentially enhancing the radionuclide concentrations in the ambient air.

7.2.3 Lathe Operator

The lathe operator would receive the highest doses from four of the nuclides that partition strongly to cast iron: Mn-54, Co-60, Ru-106+D and Ag-110m+D. His only potential exposure would be to external radiation from the cast iron in the lathe, since he would be exposed to negligible—if any—amounts of ingestible material or respirable particulates from the metal in this machine. This individual would receive higher external exposures from these nuclides than, say, the scrap cutter because the nuclides in the lathe were assumed not to be diluted. As was discussed in Chapter 5, the scrap yard is assumed to process scrap from the four nuclear plants that are scheduled for decommissioning in the same year. However, only 13% of this scrap is potentially contaminated. Thus, the scrap cutter's exposure is reduced due to the 87% uncontaminated scrap in his surroundings. The lathe, however, is assumed to come from a single furnace heat that happened to be produced from 100% potentially contaminated scrap—a reasonable assumption for an occasional occurrence. Although the radiation source is less massive, this is more than compensated by the eight-fold higher concentration.

7.2.4 EAF Furnace Operator

The EAF furnace operator would receive the highest doses from Pb-210+D because of his internal exposure to potentially contaminated dust and soot. Lead is volatile at steel-melting temperatures; however, the lead vapors condense to an aerosol dust in the cooler air outside the furnace. This dust is inhaled and ingested by the steel mill workers; when it settles and forms soot it is also inadvertently ingested. According to measured data on dust loadings at various work stations and our assumptions regarding soot ingestion rates and work assignments, the furnace operator would have the highest intake of Pb-210 of the workers modeled in our analysis. Since Pb-210 is a β-emitter with only one low-intensity, low-energy γ-ray, external exposure would not be a significant pathway.

7.3 EVALUATION OF THE RESULTS OF THE RADIOLOGICAL ASSESSMENT

The analysis was designed to produce a conservative but reasonable assessment of the potential doses and accompanying risks to individuals resulting from the recycling of steel scrap from nuclear facilities. This assessment required many assumptions regarding the scenarios and the physical processes involved. Several of the assumptions that had a significant effect on the results are discussed in the following sections.

7.3.1 Dilution of Potentially Contaminated Steel Scrap[2]

Perhaps the most critical assumptions relate to the dilution of the potentially contaminated steel scrap by uncontaminated scrap during and after recycling. Relatively little potentially contaminated steel scrap is being currently released for unrestricted recycling. Once large-scale decommissioning of nuclear facilities takes places, it is difficult to predict how much scrap will in fact be released for recycling, over what period, and with what geographic distribution. The present analysis made a conservative assumption regarding the maximum likely fraction of contaminated scrap in the process materials. Insufficient data is available to determine the

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[2] See Appendix G for a comprehensive discussion of the dilution of potentially contaminated scrap.

probability that the decommissioning scrap from four nuclear plants will be sent to the same scrap dealer in one year, and that this material will comprise the dealer's entire inventory during that year. However, due to the plants' geographical proximity to a steel mill (and thus presumably to a scrap dealer who supplies the mill) and the fact that all four plants are operated by the same electrical utility, the first assumption is reasonable. Since data on the distribution and processing capacity of scrap dealers in that geographical area was not obtainable at the time of this analysis, the assumption regarding the scrap processing capacity was the only prudent choice to be made.

The choice of the reference steel mill was made in a similar manner. Although it is based on a currently operating mill near the four nuclear plants, there is no way of assessing the probability of the reference mill being the recipient of this scrap.

The assumption regarding the lathe being made entirely of potentially contaminated steel scrap is conservative but reasonable. Although the *average* fraction of potentially contaminated steel scrap in the decommissioning scrap from the four nuclear plants is estimated to be 13%, large portions of the scrap—the primary coolant piping, for instance—consist entirely of scrap from service in an environment contaminated with radioactivity. It is reasonable to assume that one or more rail cars or trucks in convoy loaded with potentially contaminated steel scrap would arrive at an iron foundry, so that one or more heats would be fed entirely by contaminated scrap. The 8-ton lathe could be made from the metal produced from this material.

7.3.2 Exposure Pathways

A number of assumptions were made in modeling the exposure pathways for each scenario. These will be discussed separately for each pathway.

External Exposure

The external exposures were modeled using MicroShield™ 4.2 for all but two of the scenarios. MicroShield™ is an industry-standard shielding code and produces reliable results for nuclides with principal γ-ray energies greater than 100 keV. Although it is less reliable for assessing exposures from low-energy γ-emitters, this is not a serious drawback in the current analysis. The nuclides that fall into that category will have their primary impact via the inhalation pathway, so that any inaccuracy in assessing the external exposures would have little effect on the limiting doses listed in Table 7-1.

FGR 12 (Eckerman 93) provides a highly accurate means of assessing the external exposure from an idealized source geometry, if the receptor is a person standing on the source and the source has the same elemental composition as the soil used in the FGR 12 dose calculations. FGR 12 gives a reasonable approximation to the three scenarios—the slag storage yard, the road built with slag and the scrap yard—to which it was applied. In both cases, the roughness of the surface would tend to reduce the actual exposures from the FGR 12 predictions, as would the higher effective atomic number of the source material.[3]

Inhalation

The major parameters that affect the dose via the inhalation pathway are the atmospheric concentration (dust loading) and composition of the dust. The dust loading was, in most cases, based on measured values for similar operations. Thus, the dust in the areas occupied by the steel mill crane operator, the furnace operator and the operator of the continuous caster were based on reported measurements for such workers at an actual EAF steel-making facility, albeit one that primarily produced stainless steel. Since only analyses for toxic trace constituents in the dust had been performed, it was not possible to ascertain the origin of the dust, which would enable a determination of its hypothetical radioactive contamination. It was therefore assumed that it came from the furnace emissions—i.e., it had the same composition as baghouse dust. Since the furnaces are the primary source of airborne emissions in a steel mill, this is a reasonable assumption.

The dust loading in the scrap yard was more difficult to determine, since scrap processors do not routinely monitor dust levels. Newton *et al.* reported that cutting metal with an oxy-acetylene torch produces aerosol concentrations of 15 mg/m^3 (Newton 87) and that these particles were primarily of respirable size. Thus, a concentration of 15 mg/m^3 and a respirable fraction equal to 1 is an upper limit for the scrap cutting operation. However, the cutter works outdoors, so the aerosols from his torch would have more of a chance to disperse. It was thus assumed that the total dust is equal to the <u>ACGIH</u> Threshold Limit Value (TLV) of 10 mg/m^3 for nuisance dust,

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[3] A discussion of the anticipated effect of atomic number on calculations using the FGR 12 dose coefficients can be found in Section H.2.1 of Appendix H.

which is also the OSHA PEL. This is a key assumption, inasmuch as dust inhalation by the scrap cutter is the major contributor to the maximum annual dose from several radionuclides.

Another key assumption in the scrap cutting scenario is that the dust has the same specific activity as the scrap. An argument could be made that much of the radioactive contamination will be on the surface and that it is these surface layers that are the primary sources of the dust. To counter that argument, we observe that the scrap would have undergone surface decontamination prior to being released, so that loose surface activity would have been removed. The cutting operation is assumed to be the major source of the dust. The aerosols are produced by the melting and volatilization of the steel; their composition can therefore be assumed to be the same as the overall composition of the scrap.

Ingestion

Ingestion of radionuclides, either by inadvertently ingesting contaminated dust or soot during the working day or by eating food contaminated by radioactive cookware, proved to be a major contributor to the maximum doses of two of the RMEI. One is the furnace operator exposed to Pb-210. The soot ingestion rate of this worker is based on one-half of the high-end value for soil ingestion for outdoor workers listed in the EPA Exposure Factors Handbook (EPA 89), and on observations of the highly dusty, sooty environment of the steel mill. Thus, the actual ingestion rate is not likely to be more than twice what was postulated, nor is it likely to be very much lower. The soot was assumed to have the same composition as baghouse dust, since, like the dust in the air, its primary source is the fugitive emissions from the furnace. (See discussion of the inhalation pathway, above).

Ingestion is the only pathway for the RMEI exposed to Sr-90—the person whose drinking water well may become contaminated by leachate from the slag pile. This analysis includes a number of conservative assumptions, which are discussed in this section.

The first assumption regards the leachability of strontium from slag. The leach rate was calculated using diffusion coefficients which were in turn calculated using preliminary data from on-going EPA-sponsored experimental research being conducted by the Brookhaven National Laboratory. One source of uncertainty in the experimental data—the diffusion coefficients calculated from measurements on three EAF slag samples showed that the highest of the three values, which was conservatively adopted for the present analysis, was four times greater than the lowest value. Another is the assumed size of slag particles. Our calculations assumed an average particle size of 1 cm. If the slag consisted primarily of finer particles, the leach rate would be correspondingly larger.[4]

Another assumption concerned the K_d of the soil layer under the slag. For all elements considered in this study, the lowest reasonable K_d's, which had been identified in EPA 94, were used to model the transport of the radionuclides through the soil. The metallic elements (including strontium) are more mobile in an acidic environment. Slag, however, is basic. Thus, the leachate would be basic, causing elements like strontium to be retarded in the soil. This would prolong its migration time, leading to more radioactive decay of Sr-90 and perhaps even preventing any significant amount from reaching the aquifer. In such a case, of course, another scenario would result in the maximum dose.

7.3.3 Mass Distribution and Partitioning of Contaminants

The mass distribution of metal and non-metallic components among the steel or cast iron, slag, dust and home scrap was determined from a definitive study of the literature and consultations with other research workers and technical experts. The data on partitioning of contaminants among these various media was less definitive. Nevertheless, a major and largely successful effort was made to combine the observed partitioning with thermodynamic calculations to produce a set of reasonable and defensible concentration factors. The only conscious conservatism that was introduced into this phase of the analysis was the simultaneous use of high-end partition fractions for two or more media, which, as was discussed at the end of Section 6.2, would overstate the exposure of the operator of the continuous caster to four of the radionuclides studied. Since this individual did not prove to be the RMEI for any nuclide, this assumption has no effect on the maximum doses listed in Table 7-1.

7.3.4 Scenario Selection

The scenarios used in the present analysis were selected from a much longer list which had been examined in an earlier analysis of recycling residually radioactive steel scrap (SCA 95).

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[4] A more detailed discussion of the uncertainties in the slag leaching scenario can be found in Appendix L.

Scenarios in the previous analysis which were redundant or which had no potential for producing the maximum doses from any nuclide were dropped from the analysis of the RMEI. (Some of these were included in the study of the collective impacts described in Chapter 9). Given the conservative assumptions used, it is improbable that any plausible scenario would produce greater impacts than those studied.

One scenario that was not part of the assessment of the RMEI, but which was included in the assessment of collective impacts, was the use of slag as a soil conditioning agent. A scoping analysis was performed to determine if this would be a significant exposure pathway for any of the radionuclides considered in the present analysis. A brief description of this assessment is presented below. A more detailed discussion can be found in Appendix H.

Because of its high lime (CaO) content, slag is sometimes used to raise the pH of acidic soils. According to a vendor of gardening supplies, even highly acidic soils do not require more than about 100 lb of liming agent per 1,000 ft^2. The liming agent supplied by this vendor contained 48% lime, which is comparable to the CaO content of steel slags listed in Appendix I. Assuming that slag were applied to agricultural soil in the same quantity, and that it were mixed into the top six inches of soil (the assumed plow depth), the doses from the consumption of agricultural products grown in this soil were calculated.

The normalized annual dose to the RMEI via the food ingestion pathway was calculated for each radionuclide that partitions to the slag by using the dose factors calculated for the agricultural produce pathway for a generic site with radioactively contaminated soil, as listed in Table 3-1 of EPA 94. The doses calculated for this scenario for each radionuclide that would partition to the slag were one to three orders of magnitude less than the doses to the RMEI for that nuclide.

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CHAPTER 8

DETECTION AND MEASUREMENT OF CONTAMINATION

8.1 STATEMENT OF PURPOSE

The purpose of this chapter is to address issues relating to the ability to detect and quantify contamination levels of residually-contaminated scrap metal intended for recycling and in newly produced steel containing residually-contaminated scrap that was recycled. Of primary concern are questions regarding the probability of:

(1) accepting scrap metal for recycling that is assumed to be non-contaminated when in fact it is;

(2) accepting scrap that contains residual contamination well in excess of acceptable release limits;

(3) rejecting scrap metal that is falsely assumed to contain radioactivity in excess of acceptable limits, and

(4) unknowingly releasing steel derived from recycled scrap for unrestricted use (i.e., consumer products) at levels of contamination that exceed free-release criteria.

At a minimum, these complex issues require an understanding of the nature of radioactive contamination, current applicable release limits, release survey measurement methods, and limitations imposed by counting statistics, instrumentation, and radioanalytical and radiochemical analyses. These and other topics are briefly discussed in this chapter.

8.2 GUIDELINES AND STANDARDS FOR FREE RELEASE OF SCRAP ESTABLISHED BY THE NRC AND DOE

Radioactive contamination of scrap metals from commercial nuclear power plants and DOE facilities that are potentially available for recycling primarily exists in the form of surface contamination, which is limited to a few-microns-thick layer of radioactivity on the metal surface. Surfaces may be classified as "internal" and "external." For a considerably smaller fraction of potentially releasable scrap metal, the contamination may exist as bulk or volumetric

(1) Contaminated External Surfaces - Plant surfaces become contaminated over the lifetime of the plant from leaks, spills and airborne migration of radionuclides. The specific activity is low. However, the contamination is widespread and easily mobilized.

(2) Contaminated Internal Surfaces - Activated corrosion and fission products from the fuel travel through the reactor coolant water system throughout the radioactive liquid system in the plant. A portion forms a hard metallic oxide scale on the inside surfaces of pipes and equipment. This contamination is not easily mobilized.

(3) Activated Stainless Steel - Reactor internals, composed of Type 304 stainless steel, become activated by neutrons from the core. Radionuclides have very high specific activities and are immobilized inside the corrosion-resistant metal.

(4) Activated Carbon Steel - The reactor pressure vessels are made of SA533 carbon steel which becomes activated by neutrons bombardment. The specific activities are considerably lower than in the stainless steel internals. However, the binding matrix is much less corrosion resistant.

8.2.1 NRC: Regulatory Guide 1.86 (1974 and 1982)

Criteria for residual contamination levels used to decommission facilities for unrestricted use have been based on interim guidance contained in Regulatory Guide 1.86, "Termination of Operating Licenses for Nuclear Reactors," first published in June 1974, for surface contamination plus case-by-case considerations for direct radiation. Limit values from Regulatory Guide 1.86 (1974) and (1982) are shown in Table 8-1. The only significant difference between the 1.86 (1974) table and the NRC (1982) table is that the NRC (1982) table has a footnote "f," which states:

> "The average and maximum radiation levels associated with surface contamination resulting from beta-gamma emitters should not exceed 0.2 mrad/hr at 1 cm and 1 mrad/hr at 1 cm, respectively, measured through not more than 7 milligrams per square centimeter of total absorber."

Table 8-1 Regulatory Guide 1.86 Acceptable Surface Contamination Levels

Nuclides[a]	Average [b],[c],[f]	Maximum[b],[d],[f]	Removable[b],[e],[f]
U-nat, U-235, U-238, and associated decay products	5,000 dpm α per 100 cm^2	15,000 dpm α per 100 cm^2	1,000 dpm α per 100 cm^2
Transuranics, Ra-226, Ra-228, Th-230, Th-228, Pa-231, Ac-227, I-125, I-129	100 dpm per 100 cm^2	300 dpm per 100 cm^2	20 dpm per 100 cm^2
Th-nat, Th-232, Sr-90, Ra-223, Ra-224, U-232, I-126, I-131, I-133	1,000 dpm per 100 cm^2	3,000 dpm per 100 cm^2	200 dpm per 100 cm^2
Beta-gamma emitters (nuclides with decay modes other than alpha emission or spontaneous fission) except Sr-90 and others noted above.	5,000 dpm ßγ per 100 cm^2	15,000 dpm ßγ per 100 cm^2	1,000 dpm ßγ per 100 cm^2

[a] Where surface contamination by both alpha and beta-gamma emitting nuclides exist, the limits established for alpha and beta-gamma emitting nuclides should apply independently.

[b] As used in this table, dpm (disintegrations per minute) means the rate of emission by radioactive material as determined by correcting the counts per minute observed by an appropriate detector for background, efficiency, and geometric factors associated with the instrumentation.

[c] Measurements of average contaminant should not be averaged over more than 1 square meter. For objects of less surface area, the average should be derived for each such object.

[d] The maximum contamination level applies to an area of not more than 100 cm^2.

[e] The amount of removable radioactive material per 100 cm^2 of surface area should be determined by wiping that area with dry filter or soft absorbent paper, applying moderate pressure, and assessing the amount of radioactive material on the wipe with an appropriate instrument of known efficiency. When removable contamination on objects of less surface area is determined, the pertinent levels should be reduced proportionally and the entire surface should be wiped.

[f] The average and maximum radiation levels associated with surface contamination resulting from beta-gamma emitters should not exceed 0.2 mrad/hr at 1 cm and 1 mrad/hr at 1 cm, respectively, measured through not more than 7 milligrams per square centimeter of total absorber.

8.2.2 DOE Order 5400.5.

DOE's basic radiation protection standards for cleanup of residual material that include release of property are defined in Chapter IV of DOE Order 5400.5. Paragraph 4.d. states that:

> "The generic surface contamination guidelines provided in Figure IV-1 are applicable to existing structures and equipment. These guidelines are generally consistent with standards of the NRC (NRC 1982) and functionally equivalent to Section 4, 'Decontamination for Release for Unrestricted Use' of Regulatory Guide 1.86, but apply to nonreactor facilities. These limits apply to both interior equipment and building components that are potentially salvageable or recoverable scrap."

DOE release limits are, therefore, nearly identical to those of Regulatory Guide 1.86, inclusive of the NRC 1982 footnote on average and maximum dose rates of 0.2 and 1.0 mrad/hr at 1 cm. For the TRU grouping, however, DOE has held in reserve surface contamination values pending the development of standards more applicable to DOE facilities. Another difference is DOE's residual limit for external gamma radiation. Section 4.C. of Chapter IV, DOE Order 5400.4 states that:

> ". . . the average level of gamma radiation inside a building or habitable structure on a site to be released without restriction shall not exceed the background level by more than 20 μR/h."

8.2.3 Release Criteria for Volumetric Contaminants

Currently, neither the NRC or the DOE have established guidance or criteria with respect to the release of volumetrically contaminated or activated metals. DOE Order 5400.5, Chapter II, Section 5.C.(6) states:

> "No guidance is currently available for release of material that has been contaminated in depth, such as activated material or smelted contaminated metals (e.g., radioactivity per unit volume or per unit mass)."

Such material may be released based on ad hoc criteria and survey techniques approved by the DOE Office of Environment, Safety, and Health (EH).

For the NRC, volumetric release criteria had been proposed under its 1992 Below Regulatory Concern (BRC) policy. The BRC policy proposed that the average dose to individuals should be less than 10 mrem per year for each exempted practice. In addition, an interim dose criterion of 1 mrem per year average annual dose to individuals had been proposed for those practices involving widespread distribution of radioactive materials in items such as consumer products or recycled material and equipment. While NRC's BRC policy has subsequently been withdrawn, they have issued volumetric release criteria on a case-by-case basis.

8.3 VERIFICATION OF RESIDUAL CONTAMINATION OF MATERIALS RELEASED FOR UNRESTRICTED USE

Both DOE and NRC require that all materials prior to release must be surveyed to determine whether both removable and total surface contamination is less than specified limits. DOE and NRC consider material to be potentially contaminated if it has been used or stored in radiation areas that could contain unconfined radioactive material or that are exposed to beams of particles capable of causing activation (i.e., neutrons, protons, etc). Surfaces of potentially contaminated property must be surveyed using instruments and techniques appropriate for detecting stated limits.

Standard instrumentation used for a final release survey are broadly categorized as two types: portable field instruments and laboratory instruments. In general, field instruments are employed to assess total surface contamination (i.e., fixed and removable) and are employed in two discrete modes. The first involves "fixed point" or "direct" measurements, in which the detector is held in a defined stationary position above the surface for a preselected time period. Direct measurements do not attempt to evaluate the entire surface but provide a sampling of surface contamination levels. For relatively large surface areas of suspected contamination, direct measurements are taken at systematic locations to supplement scan surveys. These measurements are collected according to a predetermined pattern without regard to radiation level. Judgmental direct measurements (i.e., not chosen on a random or systematic basis) may be collected at locations where anomalous radiation levels are observed during scan surveys or at locations suspected of being contaminated.

The second mode of operating field instruments involve surface scans or "frisks." In this survey, the detector is passed slowly over the surface at a standard scanning speed while maintaining a constant distance above the surface. Since direct measurements have a low probability of identifying small areas of elevated activity, scan surveys are performed to locate such small areas, which typically represent a very small portion of the item being surveyed. In this survey, the detector is passed slowly over the surface at a standard scanning speed while maintaining a constant distance above the surface. Additionally, scan surveys may be performed to determine general contamination levels over large areas.

Laboratory instrumentation provides complementary data to field survey data. Wipe or "smear" samples provide an acceptable method for demonstrating the level of residual surface contamination that is removable. To assess volumetric sources of contamination, a portion of the potentially contaminated medium may be collected and analyzed using both chemical and instrument techniques to quantify the low levels expected in samples. Laboratory analyses can also identify individual radionuclides and establish their relative abundance when a number of contaminants are present in a mixture.

8.3.1 Total Alpha and Beta-Gamma Direct Measurements

Total alpha contamination measurements are typically performed using a zinc sulfide (ZnS) alpha scintillation detector or a thin-window gas proportional (GP) counter with a portable scaler. The probe is held at a fixed location and a 30 to 60 second count is taken. The distance between the surface and the probe is maintained at approximately 0.5 cm to 1 cm. The efficiency of the typical ZnS and GP alpha instrument is between 13% and 22%, which yields an average efficiency of about 17% for ZnS and 19% for GP counters. Expected background count rates range between 0 and 4 cpm for both detectors. The window area of the standard ZnS probe is 59 cm^2 while that of the GP detector is 126 cm^2.

Total beta contamination measurements are likely to be performed using a thin-window (1.4 to 2 mg/cm^2) "pancake" Geiger-Mueller (GM) detector or a GP counter with a portable scaler. The probe is held at a fixed location and a 30 to 60 second count is taken. Beta counting efficiencies for these instruments are extremely energy dependent, ranging from 35 - 40% at higher energies (>1000 keV maximum beta energy) to less than 5% as energies drop below 100 keV. The window area of the standard pancake probe is 20 cm^2 while that of the GP detector is 126 cm^2. Background count rates for these detectors may be as low as 50 cpm and 350 cpm, respectively; however, background may increase depending on ambient radiation fields and/or contamination levels.

8.3.2 Surface Scanning for Total Alpha and Total Beta Contamination

The term scan survey is used to describe the process of moving portable radiation detectors across a suspect surface with the intent of locating areas of contamination. The detectors used in surface scanning are identical to those used for direct measurements; however, in the scanning mode, the instrument is equipped with a ratemeter that typically has established time constant(s) settings of 2 and 10 seconds. The probe is passed slowly over the surface at scanning speeds of 5 to 15 cm per second at a relatively constant distance of 1 cm or less. The probability of detecting localized areas of contamination using scan surveys is not only affected by the sensitivity of the survey instrumentation, but also by the surveyor's ability. Personnel conducting scan surveys must interpret the audible output or visual reading of a portable survey instrument to determine if the signal being heard or seen exceeds the background level by a sufficient margin to conclude that contamination is present.

8.3.3 Surveys for Total Gamma Contamination

Sodium iodide, NaI(Tl), scintillation detectors are normally used for scanning areas for gamma emitters because they are very sensitive to gamma radiation, easily portable, and relatively inexpensive. The detector is held close to the surface (~6 cm) and moved in a serpentine (snake like, "S" shaped) pattern while walking at a speed which allows the investigator to detect the desired investigation level. A scan rate of approximately 0.5 m/s is typically used for distributed (large area) gamma emitting contamination; however, this value must be adjusted depending on the expected detector response and the desired investigation level. When surveying for small elevated areas of contamination, much slower scan rates will be required. NaI may also be used to perform direct measurements. Direct monitoring for alpha- and beta-emitters will usually result in better MDCs than gamma monitoring. However, for radionuclides that decay by electron capture, gamma monitoring is the only viable survey method since, in most cases, no betas are emitted.

The most sensitive (highest efficiency) portable detector is a 2-inch diameter (surface area - 20 cm^2) by 2-inch thick NaI. Specialized detectors are also available that optimize detection of low-energy (below 100 keV) gamma and x-radiation. Detector efficiency is largely a function of gamma energy. A detector's intrinsic efficiency is a measure of the number of counts produced relative to the number of photons incident upon the detector's surface. Intrinsic efficiency is dependent upon the photon energy, detector wall thickness, and the thickness of the scintillator. The source-detector geometry permits calculation of the fluence per unit source strength, based upon the size of the source and the distance to the detector. The combination of intrinsic efficiency, fluence per unit source strength, and detector surface area will determine the overall detector efficiency.

Background count rates with NaI detectors are dependent upon ambient background dose rate, the size of the detector, and the ratemeter/scaler setup. More advanced ratemeter/scalers permit use of threshold and window settings, which permit use of the NaI to count in the specific energy region of interest and reduce low-energy "noise." Generally, background count rates are in the range of 3,000 to 12,000 cpm, depending upon detector size.

8.4 LOWER LIMIT OF DETECTION AND MINIMUM DETECTABLE CONCENTRATION

Survey measurements of scrap metal for unrestricted release are not without potential error. There are two types of errors:

- Type I Error is the error made in assessing that contamination exists, when in fact it does not exist (false detection).
- Type II Error is the error made in concluding that contamination dose not exist, when in fact it does (false non-detection).

For fixed-point measurements, potential errors may arise due to the fact that residual contamination may be variable or localized to small areas. Thus, direct measurements and wipe samples have a statistical probability of measuring contamination levels that are not representative of overall contamination levels. A more significant contribution to Type I and II errors is the random nature of radioactive decay and the interference of background radiation levels.

<u>The Significance of LLD and MDC Values in Survey Measurements</u>. The lower limit of detection (LLD) is the smallest amount of sample activity that will yield a net count for which there is a confidence at a predetermined level that activity is present. The LLD is an *a priori* estimate of the detection capabilities of a given measurement process. It is related to the characteristics of the counting instrument, and is not dependent on other factors involved in the

measurement method or on the sample characteristics. For this reason, a statistical value called the minimum detectable concentration (MDC) has been introduced. The MDC is a level of activity concentration which is practically achievable by an overall measurement method. As distinguished from the LLD, the MDC considers not only the instrument characteristics (background and efficiency), but all other factors and conditions which influence the measurement. It is an *a priori* estimate of the activity concentration that can be practically achieved under a specified set of typical measurement conditions. These include the sample size or detector area, counting time, self-absorption and decay corrections, chemical yield, and any other factors that comprise the activity concentration determination.

MDC values are commonly based on both Type I and Type II errors by specifying a count rate for which the presence of contamination has a probability p and the probability of falsely concluding its presence is defined by (1 - p). Both NRC and DOE recommend that MDC levels be determined by setting the risks of false detection and false non-detection equal, to accept a 5% chance of incorrectly detecting activity when it is absent, and a 95% confidence that activity will be detected when it is present. Alternatively, the MDC may be considered to represent the smallest concentration of radioactive material in a sample that will be detected with 95% probability, with a 5% probability of falsely concluding that contamination is present when in fact it is not.

A third factor that profoundly affects Type I and Type II errors is the mode of instrument operation. This is especially true for instrumentation used in a scanning mode. For survey measurements that involve scanning a surface, MDC value are dictated by the complex interrelationship of (1) background levels, (2) source-detector geometry, (3) instrument time constant, (4) variability of surface contamination relative to detector surface area, and (5) scanning velocity.

8.4.1 MDCs for Surface Scanning for Small Areas of Contamination

The MDC for detection of small areas of contamination using surface scanning is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{W}{60 * V}}}{\frac{W}{60 * V} * (\sum Y_i * \varepsilon_i) * \frac{A}{100} * HF} \qquad \text{(Eq. 8-1)}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
W = detector width (cm)
60 = conversion factor (s/min)
V = detector scan rate (cm/s)
Y_i = yield for emission I (ptcle-emitted/d)
ϵ_i = detector efficiency for emission I (c/ptcle-emitted)
A = detector area (cm^2)
HF = surveyor efficiency (%).

8.4.2 MDC for Surface Scanning for Large Areas of Contamination

The MDC for detection of large areas of contamination using surface scanning is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{2 * \tau}{60}}}{\frac{2 * \tau}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100} * HF} \qquad \text{(Eq. 8-2)}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
τ = meter time constant (s)

60 = conversion factor (s/min)
Y_i = yield for emission I (ptcle-emitted/d)
ϵ_i = detector efficiency for emission I (c/ptcle-emitted)
A = detector area (cm^2)
HF = surveyor efficiency (%).

8.4.3 MDC for Direct Measurements

The MDC for direct measurements is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{t}{60}}}{\frac{t}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100}} \qquad \text{(Eq. 8-3)}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
t = count time (s)
60 = conversion factor (s/min)
Y_i = yield for emission I (ptcle-emitted/d)
ϵ_i = detector efficiency for emission I (c/ptcle-emitted)
A = detector area (cm^2).

8.5 RADIONUCLIDE MDCS FOR SURFACE CONTAMINATION

MDCs have been calculated for 40 radionuclides for surface scanning of small areas of contamination, surface scanning of large areas of contamination, and direct measurements (for a counting time of one minute). For the small area scan surveys, the MDCs represent the lowest level of a small but elevated area of surface contamination (e.g., "hot spot") that is likely to be detected with a high degree of confidence. For the large area scan surveys, the MDCs represent the lowest level of distributed sources of surface contamination that are likely to be detected.

The MDCs for direct measurements represent the sensitivity of the detection process at the location at which the survey is taken.

These MDCs should be considered as realistic, but not conservative. Background count rates, detector scan rates, and instrument counting efficiencies were chosen as relatively low, slow, and high values respectively, typical of what may be encountered under controlled, laboratory conditions. In the field, the background count rate may vary widely, scan rates will vary from individual to individual and day to day, and instrument efficiencies may be reduced by a number of factors including the source-detector geometry, source-to-detector distance, the physical condition of the surface being monitored, and surface coatings such as paint, dust, oil, or water. The MDCs for scan surveys have also been adjusted to account for the surveyor's ability in deciding whether the instrument response represents only background activity, or whether it represents residual contamination in excess of background.

MDCs have been calculated for two common beta monitoring instruments, GP and GM detectors, two alpha monitoring instruments, GP and ZnS detectors, and one gamma instrument, a NaI(Tl) detector. Detector sizes and areas were selected to represent commercially available instruments. Efficiencies represent average values compiled from historical instrument calibration data, and should be considered as the ideal efficiencies obtained under laboratory conditions.

Tables 8-2, 8-3, and 8-4 present a comparison of MDCs for small area scan surveys, large area scan surveys, and direct measurements with the limits given in NRC Regulatory Guide 1.86. The NRC's maximum surface contamination limits are used for the small area survey, while the average limits were used for the large area scan and the direct measurements. For small area surface scanning, 22 of the 40 radionuclides listed were detectable. Radionuclides which Regulatory Guide 1.86 places in group 2 (transuranics) are typically not detectable due to the 300 dpm/100cm^2 limit. Most of the radionuclides with a 15,000 dpm/100cm^2 limit were detectable. These include the alpha-emitting uranium isotopes, as well as the higher energy beta-gamma emitting radionuclides such as Co-60, Cs-134, Cs-137+D, Sr-90+D, and Tc-99. Tables 8-3 and 8-4 demonstrate the influence of longer counting times in the large area surface scans and direct measurement techniques. In these cases, 35 and 36 of the 40 radionuclides were detectable compared to the Regulatory Guide 1.86 limits. The radionuclides not detectable were the very low energy beta emitters due to their low counting efficiency.

These results indicate that, with respect to Regulatory Guide 1.86 limits, the surface scanning technique can detect small elevated areas of contamination only for radionuclides with a 15,000 dpm/100cm^2 limit. However, most radionuclides are detectable when monitoring large areas of contamination using surface scanning as well as direct measurement, even those radionuclides with a Regulatory Guide 1.86 limit as low as 100 dpm/100cm^2.

Chapter 7 of this Technical Support Document discusses the derivation of normalized doses for individuals who may be exposed to radiation as a result of the recycling of scrap. Based upon these normalized doses, surface concentration limits (in dpm/100 cm^2) have been derived for annual doses of 15, 1, and 0.1 mrem. The most challenging survey condition is the ability to identify small areas of contamination by surface scans. Table 8-5 presents a comparison of radionuclide MDCs for small area scan surveys with these derived concentration limits (DCLs) for these three dose values. For a dose limit of 15 mrem/y, all but one radionuclide is detectable, and even at 1 mrem/y, most radionuclides (31 of 40) can be detected. The situation reverses itself at 0.1 mrem/y as only 11 of 40 radionuclides are detectable.

Tables 8-6 and 8-7 compare DCLs with radionuclide MDCs for large area scan surveys and direct measurements at the three dose values. The data in these tables demonstrate that all radionuclides listed can be detected at the 15 mrem/y and 1 mrem/y dose levels. Even at 0.1 mrem/y, most radionuclides are detectable (27 of 40 with large area scans and 35 of 40 with direct measurements).

It should be reiterated that these results are based upon parameter values that represent optimal monitoring conditions. Variability of the parameters used in MDC calculations can contribute individually as well as collectively to cause an inherent variability in MDCs likely to be encountered under field conditions. Variability of these factors will cause an increase in MDC since the MDCs were calculated to represent optimal values, representative of laboratory conditions.

The factors having the most impact upon MDCs are background count rate, detector scan rate (for small areas of contamination), human factors efficiency (for surface scanning), source-detector geometry, source to detector distance, surface material condition, and surface coatings. The last four factors impact the counting efficiency. Under field conditions, the combined variability of these parameters could cause actual MDCs to be a factor of 10 higher than shown in Tables 8-2 through 8-7. Under such conditions, the detectability of radionuclides will be reduced. For instance, in the case of monitoring for small areas of contamination with surface scans, Table 8-5 showed that 31 of 40 radionuclides could be detected at a DCL of 1 mrem/y. If MDCs were increased by a factor of 10, only 11 of 40 radionuclides would be detected. For distributed sources of contamination, this reduction in detectability is not quite as significant. Table 8-6 shows that for 1 mrem/y, all radionuclides area detectable. If MDCs increase by a factor of 10, 27 of 40 radionuclides are detectable. The smallest reduction in detectability occurs in direct monitoring, where the number of radionuclides detectable in Table 8-7 drops from all 40 to 35 of 40.

This reduction in detectability may be offset for materials that are contaminated with multiple radionuclides. It may be possible to measure just one of these contaminants and still demonstrate compliance with dose limits for all of the contaminants present. In using one radionuclide to measure the presence of others, a sufficient number of high-sensitivity measurements, spatially separated throughout the material being surveyed, should be made to establish a consistent ratio. The advantage of surrogate methods is that both time and costs can be saved if the analysis of one radionuclide is simpler than the analysis of the other. The surrogate method should be used with caution due to the potential for shifts or variations in the radionuclide ratios. Physical or chemical differences between the radionuclides may cause the ratios to change following decontamination measures.

In dealing with mixtures of radionuclides, the DCLs for each radionuclide must be taken into consideration to account for the total dose relative to the appropriate dose limit. One method for adjusting DCLs is to modify the assumptions made during the exposure pathways modeling to account for multiple radionuclides. The DCLs shown in the tables in this chapter are based upon the assumption that no other radionuclides are present. A second method for adjusting DCLs is to use the unity rule to adjust the individual DCLs. The unity rule is satisfied if the sum of the ratios of the radionuclide concentration and its DCL is less than or equal to one.

Table 8-2

Detectability of Radionuclides (Small Area) by Surface Scan* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm^2)					RG 1.86	
	Beta		Alpha		Gamma	Maximum Limit	Detectable
	GP	GM	GP	ZnS	NaI		
Ac-227+D	440	2,500	85	320	120,000	300	Yes
Ag-110m+D	3,300	23,000	ND	ND	120,000	15,000	Yes
Am-241	4,300	78,000	420	1,600	340,000	300	No
C-14	3,900	31,000	ND	ND	ND	15,000	Yes
Ce-144+D	650	2,800	ND	ND	600,000	15,000	Yes
Cm-244	24,000	610,000	420	1,600	8.1E+06	300	No
Co-60	2,300	13,000	ND	ND	200,000	15,000	Yes
Cs-134	1,800	9,600	ND	ND	160,000	15,000	Yes
Cs-137+D	1,300	6,600	ND	ND	420,000	15,000	Yes
Eu-152	2,900	15,000	ND	ND	139,000	15,000	Yes
Fe-55	31,000	ND	ND	ND	ND	15,000	No
I-129	3,600	30,000	ND	ND	320,000	300	No
Mn-54	3.9E+06	1.5E+07	ND	ND	360,000	15,000	No
Mo-93	46,000	ND	ND	ND	1.2E+06	15,000	No
Nb-94	1,600	8,000	ND	ND	180,000	15,000	Yes
Ni-59	700,000	ND	ND	ND	ND	15,000	No
Ni-63	8,800	ND	ND	ND	ND	15,000	Yes
Np-237+D	1,100	7,000	440	1,600	160,000	300	No
Pa-231	4,600	220,000	430	1,600	860,000	300	No
Pb-210+D	960	5,600	420	1,600	3.9E+06	15,000	Yes
Pm-147	3,300	22,000	ND	ND	3.5E+06	15,000	Yes

Table 8-2

Detectability of Radionuclides (Small Area) by Surface Scan* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm^2)					RG 1.86	
	Beta		Alpha		Gamma	Maximum Limit	Detectable
	GP	GM	GP	ZnS	NaI		
Pu-238	19,000	460,000	420	1,600	8.5E+06	300	No
Pu-239	37,000	750,000	420	1,600	2.2E+06	300	No
Pu-240	19,000	370,000	420	1,600	8.7E+06	300	No
Pu-241	18,000	ND	1.7E+07	6.5E+07	6.9E+09	300	No
Pu-242	23,000	470,000	420	1,600	1.1E+07	300	No
Ra-226	77,000	390,000	420	1,600	5.0E+06	300	No
Ra-228+D	1,100	5,600	ND	ND	370,000	300	No
Ru-106+D	840	3,500	ND	ND	1.0E+06	15,000	Yes
Sb-125+D	2,100	13,000	ND	ND	280,000	15,000	Yes
Sr-90+D	550	2,300	ND	ND	8.7E+09	3,000	Yes
Tc-99	2,200	12,000	ND	ND	ND	15,000	Yes
Th-228+D	580	2,700	85	320	140,000	300	Yes
Th-229+D	400	2,200	85	320	110,000	300	Yes
Th-230	15,000	120,000	420	1,600	1.2E+07	300	No
Th-232	17,000	160,000	420	1,600	1.5E+07	3,000	Yes
U-234	16,000	180,000	420	1,600	9.9E+06	15,000	Yes
U-235+D	1,200	10,000	430	1,600	160,000	15,000	Yes
U-238+D	680	3,000	420	1,600	1.2E+06	15,000	Yes
Zn-65	120,000	600,000	ND	ND	740,000	15,000	No

* Scan rate = 1/3 detector width per second for alpha & beta, 15 cm/s for gamma

Table 8-3

Detectability of Radionuclides (Large Area) by Surface Scan* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm^2)					RG 1.86	
	Beta		Alpha		Gamma	Average Limit	Detectable
	GP	GM	GP	ZnS	NaI		
Ac-227+D	160	790	18	68	270	100	Yes
Ag-110m+D	1,200	7,300	ND	ND	270	5,000	Yes
Am-241	1,500	25,000	90	340	770	100	Yes
C-14	1,400	9,800	ND	ND	ND	5,000	Yes
Ce-144+D	230	890	ND	ND	1,400	5,000	Yes
Cm-244	8,400	193,000	89	330	19,000	100	Yes
Co-60	830	4,100	ND	ND	470	5,000	Yes
Cs-134	640	3,000	ND	ND	370	5,000	Yes
Cs-137+D	480	2,100	ND	ND	960	5,000	Yes
Eu-152	1,000	4,800	ND	ND	320	5,000	Yes
Fe-55	11,000	ND	ND	ND	ND	5,000	No
I-129	1,300	9,500	ND	ND	730	100	No
Mn-54	1.4E+06	4.8E+06	ND	ND	820	5,000	Yes
Mo-93	16,000	ND	ND	ND	2,800	5,000	Yes
Nb-94	570	2,600	ND	ND	410	5,000	Yes
Ni-59	250,000	ND	ND	ND	ND	5,000	No
Ni-63	3,100	ND	ND	ND	ND	5,000	Yes
Np-237+D	370	2,200	92	350	360	100	Yes
Pa-231	1,600	70,000	91	340	2,000	100	Yes
Pb-210+D	340	1,800	89	330	9,000	5,000	Yes
Pm-147	1,200	6,900	ND	ND	7.9E+06	5,000	Yes

Table 8-3

Detectability of Radionuclides (Large Area) by Surface Scan* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm^2)					RG 1.86	
	Beta		Alpha		Gamma	Average Limit	Detectable
	GP	GM	GP	ZnS	NaI		
Pu-238	6,800	145,000	89	340	20,000	100	Yes
Pu-239	13,000	2.4E+05	89	340	51,000	100	Yes
Pu-240	6,800	1.2E+05	89	340	20,000	100	Yes
Pu-241	6,400	ND	3.7E+06	1.4E+07	1.6E+07	100	No
Pu-242	8,300	150,000	89	340	24,000	100	Yes
Ra-226	27,000	120,000	89	330	12,000	100	Yes
Ra-228+D	380	1,800	ND	ND	860	100	No
Ru-106+D	300	1,100	ND	ND	2,400	5,000	Yes
Sb-125+D	730	4,000	ND	ND	650	5,000	Yes
Sr-90+D	190	730	ND	ND	2.0E+07	1,000	Yes
Tc-99	800	3,900	ND	ND	ND	5,000	Yes
Th-228+D	210	860	18	67	310	100	Yes
Th-229+D	140	710	18	67	250	100	Yes
Th-230	5,200	37,000	89	340	28,000	100	Yes
Th-232	6,000	50,000	89	340	34,000	1,000	Yes
U-234	5,600	57,000	89	340	23,000	5,000	Yes
U-235+D	410	3,200	91	340	360	5,000	Yes
U-238+D	240	970	89	340	2,800	5,000	Yes
Zn-65	42,000	190,000	ND	ND	1,700	5,000	Yes

* Meter time constant = 10 s

Table 8-4

Detectability of Radionuclides by Direct Count* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm²)					RG 1.86	
	Beta		Alpha		Gamma	Average	Detectable
	GP	GM	GP	ZnS	NaI	Limit	
Ac-227+D	70	260	6	18	91	100	Yes
Ag-110m+D	560	2,600	ND	ND	93	5,000	Yes
Am-241	690	8,000	32	91	270	100	Yes
C-14	620	3,200	ND	ND	ND	5,000	Yes
Ce-144+D	100	290	ND	ND	480	5,000	Yes
Cm-244	3,800	63,000	32	90	6,400	100	Yes
Co-60	370	1,300	ND	ND	160	5,000	Yes
Cs-134	290	990	ND	ND	130	5,000	Yes
Cs-137+D	210	680	ND	ND	330	5,000	Yes
Eu-152	460	1,600	ND	ND	110	5,000	Yes
Fe-55	4,900	ND	ND	ND	ND	5,000	Yes
I-129	580	3,100	ND	ND	250	100	No
Mn-54	620,000	1.6E+06	ND	ND	280	5,000	Yes
Mo-93	7,400	ND	ND	ND	970	5,000	Yes
Nb-94	260	830	ND	ND	140	5,000	Yes
Ni-59	110,000	ND	ND	ND	ND	5,000	No
Ni-63	1,400	ND	ND	ND	ND	5,000	Yes
Np-237+D	170	720	33	93	120	100	Yes
Pa-231	730	23,000	33	92	680	100	Yes
Pb-210+D	150	580	32	90	3,100	5,000	Yes
Pm-147	520	2,300	ND	ND	2.7E+06	5,000	Yes

Table 8-4

Detectability of Radionuclides by Direct Count* Relative to RG 1.86 Limits

Radionuclide	MDC (dpm/100cm²)					RG 1.86	
	Beta		Alpha		Gamma	Average Limit	Detectable
	GP	GM	GP	ZnS	NaI		
Pu-238	3,100	47,000	32	90	6,700	100	Yes
Pu-239	5,900	78,000	32	90	18,000	100	Yes
Pu-240	3,100	39,000	32	90	6,900	100	Yes
Pu-241	2,900	ND	1.3E+06	3.7E+06	5.4E+06	100	No
Pu-242	3,700	49,000	32	90	8,300	100	Yes
Ra-226	12,000	41,000	32	90	4,000	100	Yes
Ra-228+D	170	580	ND	ND	300	100	No
Ru-106+D	130	360	ND	ND	830	5,000	Yes
Sb-125+D	330	1,300	ND	ND	220	5,000	Yes
Sr-90+D	87	240	ND	ND	6.9E+06	1,000	Yes
Tc-99	360	1,300	ND	ND	ND	5,000	Yes
Th-228+D	93	280	6	18	110	100	Yes
Th-229+D	63	230	6	18	84	100	Yes
Th-230	2,300	12,000	32	90	9,700	100	Yes
Th-232	2,700	16,000	32	90	12,000	1,000	Yes
U-234	2,500	18,000	32	90	7,900	5,000	Yes
U-235+D	190	1,000	33	92	130	5,000	Yes
U-238+D	110	320	32	90	960	5,000	Yes
Zn-65	19,000	63,000	ND	ND	580	5,000	Yes

* Count time = 60 s

Table 8-5

Detectability of Radionuclides (Small Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC ($dpm/100cm^2$)					DCL ($dpm/100cm^2$)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	440	2,500	85	320	120,000	7,300	Yes	490	Yes	49	No
Ag-110m+D	3,300	23,000	ND	ND	120,000	19,000	Yes	1,200	No	120	No
Am-241	4,300	78,000	420	1,600	340,000	9,700	Yes	640	Yes	64	No
C-14	3,900	31,000	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	650	2,800	ND	ND	600,000	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	24,000	610,000	420	1,600	8.1E+06	17,000	Yes	1,200	Yes	120	No
Co-60	2,300	13,000	ND	ND	200,000	13,000	Yes	870	No	87	No
Cs-134	1,800	9,600	ND	ND	160,000	48,000	Yes	3,200	Yes	320	No
Cs-137+D	1,300	6,600	ND	ND	420,000	130,000	Yes	8,800	Yes	880	No
Eu-152	2,900	15,000	ND	ND	140,000	34,000	Yes	2,300	No	230	No
Fe-55	31,000	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	3,600	30,000	ND	ND	320,000	15,000	Yes	990	No	99	No
Mn-54	3.9E+06	1.5E+07	ND	ND	360,000	58,000	No	3,900	No	390	No
Mo-93	46,000	ND	ND	ND	1.2E+06	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	1,600	8,000	ND	ND	180,000	25,000	Yes	1, 700	Yes	170	No
Ni-59	700,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	8,800	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	1,100	7,000	440	1,600	160,000	7,700	Yes	510	Yes	51	No
Pa-231	4,600	220,000	430	1,600	860,000	4,700	Yes	310	No	31	No
Pb-210+D	960	5,600	420	1,600	3.9E+06	3,800	Yes	250	No	25	No
Pm-147	3,300	22,000	ND	ND	3.5E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes
Pu-238	19,000	460,000	420	1,600	8.5E+06	17,000	Yes	1,200	Yes	120	No
Pu-239	37,000	750,000	420	1,600	2.2E+06	16,000	Yes	1,100	Yes	110	No

Table 8-5

Detectability of Radionuclides (Small Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-240	19,000	370,000	420	1,600	8.7E+06	16,000	Yes	1,100	Yes	110	No
Pu-241	18,000	ND	1.7E+07	6.5E+07	6.9E+06	1.0E+06	Yes	67,000	Yes	6,700	No
Pu-242	23,000	470,000	420	1,600	1.1E+07	17,000	Yes	1,100	Yes	110	No
Ra-226	77,000	390,000	420	1,600	5.0E+06	19,000	Yes	1,300	Yes	130	No
Ra-228+D	1,100	5,600	ND	ND	370,000	64,000	Yes	4,300	Yes	430	No
Ru-106+D	840	3,500	ND	ND	1.0E+06	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	2,100	13,000	ND	ND	280,000	180,000	Yes	12,000	Yes	1,200	No
Sr-90+D	550	2,300	ND	ND	8.7E+09	40,000	Yes	2,600	Yes	260	No
Tc-99	2,200	12,000	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	580	2,700	85	320	140,000	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	400	2,200	85	320	110,000	13,000	Yes	900	Yes	90	Yes
Th-230	15,000	120,000	420	1,600	1.2E+07	18,000	Yes	1,200	Yes	120	No
Th-232	17,000	160,000	420	1,600	1.5E+07	4,100	Yes	280	No	28	No
U-234	16,000	180,000	420	1,600	9.9E+06	37,000	Yes	2,500	Yes	250	No
U-235+D	1,200	10,000	430	1,600	160,000	36,000	Yes	2,400	Yes	240	No
U-238+D	680	3,000	420	1,600	1.2E+06	41,000	Yes	2,700	Yes	270	No
Zn-65	120,000	600,000	ND	ND	740,000	120,000	Yes	8,100	No	810	No

* Scan rate = 1/3 detector width per second for beta & alpha, 15 cm per second for gamma

Table 8-6

Detectability of Radionuclides (Large Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC (dpm/100cm²)					DCL (dpm/100cm²)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	160	790	18	68	270	7,300	Yes	490	Yes	49	Yes
Ag-110m+D	1,200	7,300	ND	ND	270	19,000	Yes	1,200	Yes	120	No
Am-241	1,500	25,000	90	340	770	9,700	Yes	640	Yes	64	No
C-14	1,400	9,800	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	230	890	ND	ND	1,400	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	8,400	193,000	89	330	19,000	17,000	Yes	1,200	Yes	120	Yes
Co-60	830	4,100	ND	ND	470	13,000	Yes	870	Yes	87	No
Cs-134	640	3,000	ND	ND	370	48,000	Yes	3,200	Yes	320	No
Cs-137+D	480	2,100	ND	ND	960	130,000	Yes	8,800	Yes	880	Yes
Eu-152	1,000	4,800	ND	ND	320	34,000	Yes	2,300	Yes	230	No
Fe-55	11,000	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	1,300	9,500	ND	ND	730	15,000	Yes	990	Yes	99	No
Mn-54	1.4E+06	4.8E+06	ND	ND	820	58,000	Yes	3,900	Yes	390	No
Mo-93	16,000	ND	ND	ND	2,800	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	570	2,600	ND	ND	410	25,000	Yes	1, 700	Yes	170	No
Ni-59	250,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	3,100	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	370	2,200	92	350	360	7,700	Yes	510	Yes	51	No
Pa-231	1,600	70,000	91	340	2,000	4,700	Yes	310	Yes	31	No
Pb-210+D	340	1,800	89	330	9,000	3,800	Yes	250	Yes	25	No
Pm-147	1,200	6,900	ND	ND	7.9E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes
Pu-238	6,800	150,000	89	340	20,000	17,000	Yes	1,200	Yes	120	Yes
Pu-239	13,000	240,000	89	340	51,000	16,000	Yes	1,100	Yes	110	Yes

Table 8-6

Detectability of Radionuclides (Large Area) by Surface Scan* Relative to DCLs

Radionuclide	MDC (dpm/100cm²)					DCL (dpm/100cm²)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-240	6,800	120,000	89	340	20,000	16,000	Yes	1,100	Yes	110	Yes
Pu-241	6,400	ND	3.7E+06	1.4E+07	1.6E+07	1.0E+06	Yes	67,000	Yes	6,700	Yes
Pu-242	8,300	150,000	89	340	24,000	17,000	Yes	1,100	Yes	110	Yes
Ra-226	27,000	120,000	89	330	12,000	19,000	Yes	1,300	Yes	130	Yes
Ra-228+D	380	1,800	ND	ND	860	64,000	Yes	4,300	Yes	430	Yes
Ru-106+D	300	1,100	ND	ND	2,400	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	730	4,000	ND	ND	650	180,000	Yes	12,000	Yes	1,200	Yes
Sr-90+D	190	730	ND	ND	2.0E+07	40,000	Yes	2,600	Yes	260	Yes
Tc-99	800	3,900	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	210	860	18	67	310	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	140	710	18	67	250	13,000	Yes	900	Yes	90	Yes
Th-230	5,200	37,000	89	340	28,000	18,000	Yes	1,200	Yes	120	Yes
Th-232	6,000	50,000	89	340	34,000	4,100	Yes	280	Yes	28	No
U-234	5,600	57,000	89	340	23,000	37,000	Yes	2,500	Yes	250	Yes
U-235+D	410	3,200	91	340	360	36,000	Yes	2,400	Yes	240	Yes
U-238+D	240	970	89	340	2,800	41,000	Yes	2,700	Yes	270	Yes
Zn-65	42,457	190,000	ND	ND	1,700	120,000	Yes	8,100	Yes	810	No

* Meter time constant = 10 s

Table 8-7

Detectability of Radionuclides by Direct Count* Relative to DCLs

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for 15 mrem/y	Detectable	for 1 mrem/y	Detectable	for 0.1 mrem/y	Detectable
	GP	GM	GP	ZnS	NaI						
Ac-227+D	70	260	6	18	91	7,300	Yes	490	Yes	49	Yes
Ag-110m+D	560	2,600	ND	ND	93	19,000	Yes	1,200	Yes	120	Yes
Am-241	690	8,000	32	91	270	9,700	Yes	640	Yes	64	Yes
C-14	620	3,200	ND	ND	ND	1.4E+07	Yes	900,000	Yes	90,000	Yes
Ce-144+D	100	290	ND	ND	480	1.3E+06	Yes	89,000	Yes	8,900	Yes
Cm-244	3,800	63,000	32	90	6,400	17,000	Yes	1,200	Yes	120	Yes
Co-60	370	1,300	ND	ND	160	13,000	Yes	870	Yes	87	No
Cs-134	290	990	ND	ND	130	48,000	Yes	3,200	Yes	320	Yes
Cs-137+D	210	680	ND	ND	330	130,000	Yes	8,800	Yes	880	Yes
Eu-152	460	1,600	ND	ND	110	34,000	Yes	2,300	Yes	230	Yes
Fe-55	4,900	ND	ND	ND	ND	1.8E+09	Yes	1.2E+08	Yes	1.2E+07	Yes
I-129	580	3,100	ND	ND	250	15,000	Yes	990	Yes	99	No
Mn-54	620,000	1.6E+06	ND	ND	280	58,000	Yes	3,900	Yes	390	Yes
Mo-93	7,400	ND	ND	ND	970	2.1E+08	Yes	1.4E+07	Yes	1.4E+06	Yes
Nb-94	260	830	ND	ND	140	25,000	Yes	1,700	Yes	170	Yes
Ni-59	110,000	ND	ND	ND	ND	2.7E+09	Yes	1.8E+08	Yes	1.8E+07	Yes
Ni-63	1,400	ND	ND	ND	ND	1.1E+09	Yes	7.3E+07	Yes	7.3E+06	Yes
Np-237+D	170	720	33	93	120	7,700	Yes	510	Yes	51	Yes
Pa-231	730	23,000	33	92	680	4,700	Yes	310	Yes	31	No
Pb-210+D	150	580	32	90	3,100	3,800	Yes	250	Yes	25	No
Pm-147	520	2,300	ND	ND	2.7E+06	8.3E+07	Yes	5.5E+06	Yes	550,000	Yes
Pu-238	3,100	47,000	32	90	6,700	17,000	Yes	1,200	Yes	120	Yes
Pu-239	5,900	78,000	32	90	18,000	16,000	Yes	1,100	Yes	110	Yes

Table 8-7

Detectability of Radionuclides by Direct Count* Relative to DCLs

Radionuclide	MDC (dpm/100cm^2)					DCL (dpm/100cm^2)					
	Beta		Alpha		Gamma	for		for		for	
	GP	GM	GP	ZnS	NaI	15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pu-240	3,100	39,000	32	90	6,900	16,000	Yes	1,100	Yes	110	Yes
Pu-241	2,900	ND	1.3E+06	3.7E+06	5.4E+06	1.0E+06	Yes	67,000	Yes	6,700	Yes
Pu-242	3,700	49,000	32	90	8,300	17,000	Yes	1,100	Yes	110	Yes
Ra-226	12,000	41,000	32	90	4,000	19,000	Yes	1,300	Yes	130	Yes
Ra-228+D	170	580	ND	ND	300	64,000	Yes	4,300	Yes	430	Yes
Ru-106+D	130	360	ND	ND	830	460,000	Yes	30,000	Yes	3,000	Yes
Sb-125+D	330	1,300	ND	ND	220	180,000	Yes	12,000	Yes	1,200	Yes
Sr-90+D	87	240	ND	ND	6.9E+06	40,000	Yes	2,600	Yes	260	Yes
Tc-99	360	1,300	ND	ND	ND	5.5E+08	Yes	3.6E+07	Yes	3.6E+06	Yes
Th-228+D	93	280	6	18	110	44,000	Yes	2,900	Yes	290	Yes
Th-229+D	63	230	6	18	84	13,000	Yes	900	Yes	90	Yes
Th-230	2,300	12,000	32	90	9,700	18,000	Yes	1,200	Yes	120	Yes
Th-232	2,700	16,000	32	90	12,000	4,100	Yes	280	Yes	28	No
U-234	2,500	18,000	32	90	7,900	37,000	Yes	2,500	Yes	250	Yes
U-235+D	190	1,000	33	92	130	36,000	Yes	2,400	Yes	240	Yes
U-238+D	110	320	32	90	960	41,000	Yes	2,700	Yes	270	Yes
Zn-65	19,000	63,000	ND	ND	580	120,000	Yes	8,100	Yes	810	Yes

* Count time = 60 s

8.6 THE POTENTIAL IMPACT OF INTRODUCING RADIOACTIVELY CONTAMINATED METAL IN THE PRODUCTION OF STEEL

When surface contaminated scrap metal is introduced in the smelting process, the newly produced steel can be expected to be volumetrically contaminated. The potential impact of surface contaminated scrap metal on contamination levels in the metal melt can be estimated by means of the following equation:

$$C_{mm} = \frac{S_{RSM}}{th_{RSM}\ \rho_{RSM}} \ x\ mf_{RSM}\ x\ af_{mm}\ x\ 0.45 \qquad \text{(Eq. 8-4)}$$

where:

C_{mm} = concentration in metal melt (pCi/g),
S_{RSM} = surface contamination of scrap metal (dpm/100 cm^2),
th_{RSM} = thickness of scrap metal (cm),
r_{RSM} = density of scrap metal (g/cm^3),
mf_{RSM} = mass fraction of contaminated scrap metal in new steel melt,
af_{mm} = fraction of scrap metal radioactivity transferred to new steel melt,
0.45 = conversion factor (pCi/dpm).

An upper bound estimate of volumetric contamination may be derived from the following conservative assumptions:

(1) all scrap has residual surface contamination at the maximum acceptable release limit of 5,000 dpm/100 cm^2,

(2) the average thickness of scrap is 1 cm,

(3) density of scrap is 7.8 g/cm^3,

(4) newly produced steel melt is derived from 40% scrap (60% uncontaminated virgin material), and

(5) all surface radioactivity is transferred to new steel melt.

Substituting these conservative values into the above equation yields a bounding estimates for volumetric contamination in new steel melt of 1.15 pCi/g. Radionuclides for which acceptable release limits are 1,000 dpm/100 cm^2 and 100 dpm/100 cm^2 yield bounding values of 0.23 pCi/g and 0.023 pCi/g, respectively.

8.7 LIMITATIONS OF STANDARD SURVEY MEASUREMENTS WHEN SCRAP IS CONTAMINATED VOLUMETRICALLY

When scrap metal is potentially contaminated volumetrically, the limits of detection by standard release survey measurements are considerably more difficult and less sensitive. From Equation 8-4, it can be shown that the impact of recycling contaminated scrap metal with a surface contamination of 5,000 dpm/100 cm^2 is equivalent to recycling scrap metal with a volumetric contamination of 2.9 pCi/g. Any attempt to establish parity between existing release criteria for surface contamination and future criteria for volumetric contamination must, therefore, address the issue of survey measurements of volumetric contaminants. In addition to all factors which define detection limits for surface contamination, limits of detection for bulk contamination are further hampered by the short ranges of alpha and beta particles. Detection is limited to radiation emissions occurring at a depth that is less than the range of the particulate radiation.

8.7.1 Limitations When Bulk Contaminant is a Beta Emitter

In effect, such survey measurements are synonymous with counting samples of "infinite" thickness for which absorption of radiation by the "sample" occurs. An estimate of activity level per unit volume or weight, therefore, requires that the observed count rate be adjusted for sample absorption and must further assume that contaminant is distributed uniformly throughout the sample. To account for self absorption when the contaminant is a beta emitter, the following formula is used:

$$F_S = \frac{R_o}{R} = \frac{1}{\mu_M X}(1 - e^{-\mu_M X}) \qquad \text{(Eq. 8-5)}$$

where:

F_S = self absorption factor,

R_o = measured activity,
R = true activity,
x = sample thickness (mg/cm^2),
μ_M = absorption coefficient (cm^2/mg) (see NBS Handbook No. 51, pg 26).

Inspection of Equation 8-5 reveals that the self absorption factor is a function of the sample thickness and μ_M:

$$\mu_M(cm^2/mg) = \frac{0.693}{HVT(mg/cm^2)} \qquad \text{(Eq. 8-6)}$$

where HVT is the half-value thickness based on maximum beta energy.

The use of Equations 8-5 and 8-6 and the limits of sensitivity for volumetric contaminated material can be illustrated by the following example.

Sample Calculation: Calculate the volumetric activity (i.e., pCi/g) of a 1 cm thick slab of steel uniformly contaminated with the activation nuclide Co-60. A fixed-point surface measurement with a pancake probe yields a net count rate of 50 cpm.

Solution:

$$R = \frac{(R_o)\,\mu_M X}{(1 - e^{\mu_M X})}$$

where:

$$R_o = 50\,cpm \qquad \mu_M = \frac{0.693}{(HVL)_{\beta\text{-}Max}} = \frac{0.693}{8\,mg/cm^2} = 0.087\,cm^2/mg$$

$$X = \frac{\rho_{Steel}}{thickness} \approx \frac{7800\,mg/cm^3}{1\,cm} \approx 7800\,mg/cm^2$$

$$R = \frac{(50\,cpm)(675)}{1} = 33{,}783\,cpm$$

To convert count rate (cpm) to sample activity (dpm per unit volume or weight) assume that the pancake probe has an effective area of 20 cm^2 on contact and a 10% counting efficiency.

$$BulkActivity = \frac{R}{(A)(eff)(th)(\rho)(2.22)}$$

$$BulkActivity = \frac{33{,}783\,cpm}{(20\,cm^2)(0.2\,cpm/dpm)(1\,cm)(7.8\,g/cm^3)(2.22\,dpm/pCi)}$$

$$BulkActivity = 975\,pCi/g$$

Note: The selected R_o value of 50 cpm approximates the lower limits of measurable activity for the pancake probe in an environment where the background level is 100 cpm. Correspondingly, the value of 975 pCi/g also approximates the MDC value for any material volumetrically contaminated with Co-60 having a density thickness of 7,800 mg/cm^2, which, for steel, corresponds to a thickness of 1 cm.

8.7.2 Limitations When Bulk Contaminant is an Alpha Emitter

Owing to the much shorter range of alpha particles, any observed activity by standard alpha survey instruments must be assumed to have originated within the first micron of metal. This assumption can be approximated from the formula which defines the range (cm) of alpha particles in air.

$$R_\gamma = 1.24E - 2.62$$

where E is the alpha particle energy in MeV.

An alpha particle of 5 MeV is estimated to have a range of about 3.6 cm in air having a density of 0.0013 g/cm^3. For steel having a density of about 7.8 g/cm^3, the range of a 5 MeV alpha particle would correspond to less than 1 micron. Assuming that an alpha particle would have to have a minimum residual energy of 2.5 MeV in order to trigger a pulse, it can be assumed that any detected alpha radiation had to originate within about 1 micron from the surface. For bulk contamination with a uniform contamination over a thickness of 1 cm yielding a net alpha count rate of 10 cpm, the volumetric contamination can be estimated to be greater than 580 pCi/g.

Note: A net count rate of about 10 cpm with a ZnS detector operating at a 17% efficiency in a 3 cpm background environment represents the lower limit of detectable activity. Thus, standard alpha survey techniques are quite insensitive to volumetrically distributed conatamination in steel.

8.7.3 Limitations for Gamma Emitting Bulk Contaminants

Gamma radiation is not subject to the intense self absorption that limits detection of particulate radiation under conditions of volumetric contamination. It would appear, therefore, that a NaI or GeLi detector might have a suitable application in quantifying volumetrically distributed gamma-emitting contaminants in scrap metal. Realistically, however, quantitative measurements for these detection systems can only be obtained under rigidly controlled conditions in which the assessed sample is essentially identical to a standard calibration source(s) in terms of size, mass, configuration, counting geometry, and gamma energy emissions.

8.8 ASSESSING THE RADIONUCLIDE CONCENTRATION IN STEEL PRODUCED FROM SCRAP

The aforementioned difficulty and impracticality of assessing volumetric contamination in steel derived from recycled scrap are significantly reduced when such measurements are sought for the metal melt. This is due to the fact that the smelting process can be assumed to (1) convert all contaminants (surface and volumetric) into volumetric contaminants and (2) distribute contamination with relative uniformity in the metal melt. Consequently, only a limited number of samples are needed to characterize a large mass of metal melt (i.e., for a basic oxygen furnace, the mass per charge is 220 tons). Furthermore, samples may be obtained from the metal melt in the form of standard ingots, which provide a suitable basis for developing calibration standards for NaI and GeLi systems. Sub-samples of ingots may also be analyzed by radiochemical methods.

Radiochemical analysis requires the complete dissolution of the metal sample in acid(s), the chemical separation of radioisotopes belonging to a common element, and the quantification of radionuclides on the basis of their emission(s) by means of a suitable detection system.

8.9 MDCS AND ASSOCIATED PARAMETERS FOR LABORATORY ANALYSIS OF RADIONUCLIDES

The MDC for laboratory measurements is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * t}}{t * Y_i * \varepsilon_i * M * R} \qquad \text{(Eq. 8-7)}$$

where:

MDC = minimum detectable concentration (pCi/g)
B_R = detector background count rate (cpm)
t = count time (min)
Y_i = yield for emission I (ptcle/d)
ϵ_i = detector efficiency for emission I (c/ptcle)
M = sample mass (g)
R = chemical yield.

For this analysis, MDCs for laboratory analysis of solid samples were obtained from an article by F.M. Cox and C.F. Guenther (Cox 1995). The authors present a range of MDCs as reported by 24 commercial and government laboratories. The article is quite applicable to this analysis because it presents state-of-the-art MDCs and associated parameter values for laboratory analysis of radioactive materials in solids, as well as analysis costs as a function of MDC.

The MDC column contains information that integrates the effects of background levels, detection efficiencies, count time, and sample size to give a lower limit of what could reasonably be detected with a given level of confidence. The values given represent the state-of-the-art detection capabilities as reported by commercial analytical laboratories. While increasing the count time or sample size can lead to the detection of lower concentrations, it should be realized that there are practical limits on detection that are driven by time constraints, background levels, cost, and the desired level of confidence. For most radionuclides, reported background count rates were below one count per minute and yielded corresponding lower limits of detecting sample activity that were less than 1 pCi/g.

In most cases under "Mode of detection," the entry was either alpha, beta, or gamma. In several cases, one could use two different modes of detection for the same isotope. Examples, are I-129 and Pb-210 where either beta or gamma analysis could be used. In the case of Th-232, both alpha and gamma analyses are cited as acceptable. The following provides a brief overview of instrumentation commonly employed by the reporting analytical laboratories:

- Gross alpha measurements were typically made using either scintillation devices (such as ZnS in conjunction with a photomultiplier tube), gas-flow proportional detectors (both "windowless" and thin-window types), or thin-window GM tubes. Gas flow counters using P-10 gas (a 90/10% blend of argon/methane gases) were the most popular.

- For gross beta measurement, gas-flow proportional and thin-window GM tube-type detection systems were the most popular. Some advanced liquid-scintillation detectors were also used for general alpha and beta detection.

- Both gross gamma and gamma spectrometric measurements were made using the popular and very effective sodium iodide [Na(Tl)] and germanium (Ge) detector systems.

Typical background rate, B_R, includes the response of a nuclear detector to natural background radiation. Such background radiation can be either internal to the detector (as a result of natural radioactive impurities in the detector materials) or external, such as terrestrial or cosmic that penetrates into the collection region of the detector. Backgrounds of alpha radiation were typically low, about 0.1 cpm, vary diurnally with radon and progeny air concentrations and were practically invariant with the system used. Backgrounds of beta radiation varied from 10 to 30 cpm, again basically system invariant. Backgrounds of gamma radiation were totally the result of external gamma and cosmic-ray backgrounds, and hence increased with altitude/elevation and varied with building materials.

Count times, t, and sample mass, M, show the range of counting times and sample sizes necessary to achieve the MDC ranges indicated. Note that count times range from 20 minutes to almost 17 hours (1,000 minutes), while sample mass ranges from 0.1 to 750 g.

System detection efficiency, ϵ, refers to the fraction of emitted radiation from the sample that impinged upon a detector per unit time that was converted into a measurable signal, such as counts per unit time. All variables such as geometry, intrinsic efficiency, etc., were taken into account. For alpha and beta measurements, the efficiency can reasonably vary from 5% to 40%, depending on detector type. Other important factors that can affect overall system efficiency were the distance between the source (sample) and detector surface (particularly for alpha), and sample area to detector area (surface area to surface area). For example, the best geometry for detection of alpha and beta radiation was one in which the surface areas of both were the same. For gamma rays, the efficiency was always low (less than a few %), and for specific energy peaks representing specific isotopes, the value was even lower (<1%).

Chapter 7 of this Technical Support Document discusses the derivation of normalized doses for individuals who may be exposed to radiation as a result of the recycling of scrap metal from nuclear facilities. Based upon these normalized doses, volumetric concentration limits (in pCi/g) have been derived for annual doses of 15, 1, and 0.1 mrem. Table 8-7 presents a comparison of radionuclide MDCs by laboratory analysis with these derived concentration limits (DCLs) for these three dose values. These results demonstrate the excellent sensitivity that can be achieved when samples of the metal melt are obtained for laboratory analysis. Laboratory MDCs are adequate to detect all 40 radionuclides for the 15 and 1 mrem/y cases. For 0.1 mrem/y, only 6 of the radionuclides could not be detected. In each of these cases, the MDC was only slightly higher than the DCL. If deemed necessary, the laboratory could reach these DCLs by using a longer count time for samples which must be analyzed for these radionuclides. Note however (see Table 8-6) that the lowest MDCs for some of these radionuclides, particularly the alpha emitters, is already based upon count times ranging from 7 to 17 hours per sample.

Table 8-8

Laboratory MDCs, Associated Parameters, and Costs

Radionuclide	MDC (pCi/g)	Mode of detection	B_R (cpm)	t (min)	ε (%)	M (g)	Price ($/sample) to detect -		
							10.0 pCi/g	1.0 pCi/g	0.1 pCi/g
Ac-227+D	0.05 - 0.4	α	0.03	400 - 600	14 - 25	1 - 100	80 - 178	100 - 210	135 - 270
Ag-110m+D	0.04	γ	1	100 - 500	0.6 - 30	500	95 - 172	95 - 220	105 - 260
Am-241	0.05 - 0.5	α	0.008 - 0.01	400 - 700	14 - 35	1 - 10	95 - 250	110 - 300	145 - 375
C-14	0.2 - 37	β	8 - 21	25 - 200	65 - 75	0.1 - 50	40	90	100
Ce-144+D	0.23	γ	1	100 - 500	2.2	500	95 - 178	95 - 220	105 - 260
Cm-244	0.05 - 0.1	α	0.005 - 0.01	400 - 700	14 - 35	1 - 10	95 - 250	110 - 300	145 - 375
Co-60	0.01 - 0.3	γ	1	100 - 500	0.9 - 30	100 - 500	95 - 178	95 - 220	105 - 260
Cs-134	0.02 - 0.4	γ	0.2 - 1	100 - 500	0.12 - 30	100 - 500	95 - 172	95 - 220	105 - 260
Cs-137+D	0.007 - 0.3	γ	0.2 - 1	100 - 500	0.6 - 30	100 - 175	95 - 178	95 - 220	105 - 260
Eu-152	0.02 - 0.9	γ	0.08 - 1	100 - 500	0.18 - 30	100 - 500	95 - 178	95 - 220	105 - 260
Fe-55	1 - 30	β/γ	2.5 - 20	20 - 100	20 - 35	1 - 50	60 - 172	80 - 220	110 - 260
I-129	0.4 - 2	β/γ	0.35 - 2	50 - 100	0.3 - 22	500	90 - 200	100 - 220	125 - 260
Mn-54	0.2 - 0.3	γ	1	100 - 500	0.5	100 - 500	95 - 172	95 - 220	105 - 260
Mo-93	0.2 - 150	β	6 -191	25 - 200	40 - 50	0.1 - 50	40 - 110	90 - 140	100 - 170
Nb-94	0.2 - 0.3	γ	1	100 - 500	0.5	100 - 500	95 - 172	95 - 220	105 - 260
Ni-59	1 - 30	β/γ	2.5 - 20	20 - 100	20 - 35	1 - 50	60 - 172	80 - 220	110 - 260
Ni-63	1 - 100	β	20	20 - 50	30 - 50	0.1 - 50	60 - 200	75 - 220	110 - 240
Np-237+D	0.05 - 0.5	α	0.01	400 - 700	14 - 25	1 - 10	80 - 178	100 - 210	135 - 270
Pa-231	0.05 - 0.5	α	0.01	400 - 700	14 - 25	1 - 10	80 - 178	100 - 210	135 - 270
Pb-210+D	0.1 - 2	α	0.01	240 - 400	14 - 20	1 - 50	70 - 200	90 - 240	120 - 300

Table 8-8

Laboratory MDCs, Associated Parameters, and Costs

Radionuclide	MDC (pCi/g)	Mode of detection	B_R (cpm)	t (min)	ε (%)	M (g)	Price ($/sample) to detect - 10.0 pCi/g	1.0 pCi/g	0.1 pCi/g
Pm-147	0.5 - 5	β	2 - 2.5	100	20 - 40	1 - 50	70 - 172	95 - 220	120 - 260
Pu-238	0.02 - 0.4	α	0.004 - 0.01	400 - 1,000	14 - 35	1 - 10	95 - 200	110 - 200	145 - 300
Pu-239	0.02 - 0.4	α	0.004 - 0.01	400 - 1,000	14 - 35	1 - 10	95 - 200	110 - 200	145 - 300
Pu-240	0.05 - 0.4	α	0.01	400 - 1,000	14 - 35	1 - 10	95 - 200	110 - 200	145 - 300
Pu-241	0.02 - 20	β	2 - 10	100 - 600	15 - 20	1 - 10	80 - 250	90 - 300	110 - 375
Pu-242	0.02 - 0.4	α	0.004 - 0.01	400 - 1,000	14 - 35	1 - 10	95 - 200	110 - 200	145 - 300
Ra-226	0.02 - 0.7	γ	0.17 - 1	100 - 400	0.7 - 30	100 - 750	70 - 172	80 - 220	105 - 260
Ra-228+D	0.1 - 2	γ	0.21 - 1	100 - 400	0.35 - 30	100 - 750	75 - 172	90 - 220	115 - 260
Ru-106+D	0.2 - 1	γ	1	100 - 500	0.7 - 30	100 - 500	95 - 172	95 - 220	105 - 260
Sb-125+D	0.11	γ	1	100 - 500	1 - 30	500	95 - 172	95 - 220	105 - 260
Sr-90+D	0.03 - 5	β	0.5 - 1	100	20 - 58	1 - 25	75 - 222	95 - 165	125 - 225
Tc-99	0.3 - 15	β	2 - 30	50 - 100	20 - 33	1 - 10	75 - 220	95 - 240	120 - 300
Th-228+D	0.05 - 0.4	α	0.03	400 - 600	14 - 25	1 - 100	80 - 178	100 - 210	135 - 270
Th-229+D	0.05 - 0.4	α	0.03	400 - 600	14 - 25	1 - 100	80 - 178	100 - 210	135 - 270
Th-230	0.05 - 0.5	α	0.01	400 - 700	14 - 25	1 - 10	80 - 178	100 - 210	135 - 270
Th-232	0.05 - 2	α/γ	0.01	400 - 700	25	1 - 10	80 - 195	100 - 210	135 - 270
U-234	0.05 - 0.2	α	0.01	400 - 600	14 - 25	1 - 10	85 - 166	95 - 200	135 - 250
U-235+D	0.02 - 0.3	α	0.004 - 0.01	400 - 1,000	14 - 25	1 - 10	85 - 166	95 - 200	135 - 250
U-238+D	0.02 - 0.1	α	0.01	400 - 1,000	14 - 25	1 - 10	95 - 166	110 - 200	145 - 250
Zn-65	0.09 - 0.6	γ	1	100 - 500	0.4 - 30	100 - 500	95 - 172	95 - 220	105 - 260

Table 8-9

Detectability of Radionuclides by Laboratory Analysis Relative to DCLs

Radionuclide	MDC (pCi/g)	DCL (pCi/g)					
		15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Ac-227+D	0.05 - 0.4	1.9	Yes	0.13	Yes	0.013	No
Ag-110m+D	0.04	24	Yes	1.6	Yes	0.16	Yes
Am-241	0.05 - 0.5	12	Yes	0.82	Yes	0.082	Yes
C-14	0.2 - 37	17,000	Yes	1,200	Yes	120	Yes
Ce-144+D	0.23	850	Yes	57	Yes	5.7	Yes
Cm-244	0.05 - 0.1	22	Yes	1.5	Yes	0.15	Yes
Co-60	0.01 - 0.3	17	Yes	1.1	Yes	0.11	Yes
Cs-134	0.02 - 0.4	61	Yes	4.1	Yes	0.41	Yes
Cs-137+D	0.007 - 0.3	170	Yes	11	Yes	1.1	Yes
Eu-152	0.02 - 0.9	44	Yes	2.9	Yes	0.29	Yes
Fe-55	1 - 30	2.2E+06	Yes	150,000	Yes	15,000	Yes
I-129	0.4 - 2	19	Yes	1.3	Yes	0.13	No
Mn-54	0.2 - 0.3	74	Yes	5.0	Yes	0.50	Yes
Mo-93	0.2 - 150	270,000	Yes	18,000	Yes	1,800	Yes
Nb-94	0.2 - 0.3	32	Yes	2.1	Yes	0.21	Yes
Ni-59	1 - 30	3.4E+06	Yes	230,000	Yes	23,000	Yes
Ni-63	1 - 100	1.4E+06	Yes	93,000	Yes	9,300	Yes
Np-237+D	0.05 - 0.5	9.8	Yes	0.66	Yes	0.066	Yes
Pa-231	0.05 - 0.5	6.0	Yes	0.40	Yes	0.040	No
Pb-210+D	0.1 - 2	4.9	Yes	0.33	Yes	0.033	No

Table 8-9

Detectability of Radionuclides by Laboratory Analysis Relative to DCLs

Radionuclide	MDC (pCi/g)	DCL (pCi/g)					
		15 mrem/y	Detectable	1 mrem/y	Detectable	0.1 mrem/y	Detectable
Pm-147	0.5 - 5	110,000	Yes	7,000	Yes	700	Yes
Pu-238	0.02 - 0.4	22	Yes	1.5	Yes	0.15	Yes
Pu-239	0.02 - 0.4	21	Yes	1.4	Yes	0.14	Yes
Pu-240	0.05 - 0.4	21	Yes	1.4	Yes	0.14	Yes
Pu-241	0.02 - 20	1,300	Yes	85	Yes	8.5	Yes
Pu-242	0.02 - 0.4	22	Yes	1.4	Yes	0.14	Yes
Ra-226	0.02 - 0.7	24	Yes	1.6	Yes	0.16	Yes
Ra-228+D	0.1 - 2	41	Yes	2.7	Yes	0.27	Yes
Ru-106+D	0.2 - 1	290	Yes	19	Yes	1.9	Yes
Sb-125+D	0.11	240	Yes	16	Yes	1.6	Yes
Sr-90+D	0.03 - 5	25	Yes	1.7	Yes	0.17	Yes
Tc-99	0.3 - 15	700,000	Yes	46,000	Yes	4,600	Yes
Th-228+D	0.05 - 0.4	11	Yes	0.74	Yes	0.074	Yes
Th-229+D	0.05 - 0.4	3.4	Yes	0.23	Yes	0.023	No
Th-230	0.05 - 0.5	23	Yes	1.6	Yes	0.16	Yes
Th-232	0.05 - 2	5.3	Yes	0.35	Yes	0.035	No
U-234	0.05 - 0.2	48	Yes	3.2	Yes	0.32	Yes
U-235+D	0.02 - 0.3	46	Yes	3.1	Yes	0.31	Yes
U-238+D	0.02 - 0.1	52	Yes	3.5	Yes	0.35	Yes
Zn-65	0.09 - 0.6	160	Yes	10	Yes	1.0	Yes

8.10 SUMMARY

Standard scan survey techniques for small areas of contamination are only marginally adequate relative to Regulatory Guide 1.86 limits, detecting slightly more than one-half of the radionuclides. Only those radionuclides listed in group 4 of Regulatory Guide 1.86 (with a limit of 15,000 dpm/100 cm^2) can be reliability detected. Significant improvement can be obtained when surveying for distributed sources of contamination (almost 90% of the radionuclides detected), even detecting radionuclides as low as 100 dpm/100 cm^2.

Relative to the 15 mrem/y DCL, small areas of contamination can be detected for all but one radionuclide. At 1 mrem/y, detectability drops to 75%, while only 25% of the radionuclides are detectable at 0.1 mrem/y . Significant improvement is noted when surveying for distributed sources of contamination. One hundred percent of the radionuclides are detectable at both 15 mrem/y and 1 mrem/y. At 0.1 mrem/y, almost 70% of the radionuclides are detectable while scanning for large areas, while almost 90% could be detected using direct measurements.

These results should be considered to represent optimal monitoring conditions. Under field conditions, levels of detectability would increase, perhaps due to a number of factors most of which involve a loss of counting efficiency. An increase of a factor of 10 is possible above the MDCs presented in these tables. Such an increase would have a significant effect on detectability, particularly for small areas of contamination. The extent of the loss of detectability is less for scanning for large areas of contamination and for direct measurements. Surrogate methods may be useful in situations where there are multiple radionuclides present.

Any assessment of volumetrically contaminated metal by standard field survey techniques is severely restricted by the limited range of particulate radiation. Only those radionuclides with DCLs greater than a few hundred pCi/g could be detected reliably. However, laboratory analysis of samples of scrap steel or steel derived from the recycling of scrap metal provides significantly improved results. State-of-the-art laboratory methods are quite effective at detecting low levels of volumetric contamination, even down to a few tenths to even hundredths of a pCi/g. At levels corresponding to 15 mrem/yr and 1 mrem/yr, 100% of the radionuclides evaluated could be detected. Even at 0.1 mrem/yr, 85% of the radionuclides are detectable.

8.11 REFERENCES

Cox, F.M. and C.S. Guenther. 1995. *An Industry Survey of Current Lower Limits of Detection for Various Radionuclides*. Health Physics 69, 121-129.

U.S. Nuclear Regulatory Commission. 1974. *Termination of Operating Licenses for Nuclear Reactors*. NRC Regulatory Guide 1.86, Washington, D.C.

U.S. Nuclear Regulatory Commission. 1982. *Termination of Operating Licenses for Nuclear Reactors*. NRC Regulatory Guide 1.86 (Draft), Washington, D.C.

CHAPTER 9

NORMALIZED COLLECTIVE IMPACTS MODELS

This chapter describes the development of total normalized collective doses and risks for each of the radionuclides considered. These normalized doses and risks can then be used with estimates of the mass of metal to be recycled, and its radionuclide concentrations, to calculate the total collective dose and risk received by humans due to the recycle of scrap metal from nuclear facilities. The environmental pathways that are major contributors to the collective dose and risk are described in this chapter and exposure models are developed. Figure 9-1 presents a simplified schematic diagram of some of the potential exposure scenarios that could result from recycling scrap metal from nuclear facilities into consumer and/or industrial products. In general, the exposure scenarios can be broken into five categories: air emissions, slag, baghouse dust, steel (these four categories are associated with the melting process) and transportation.

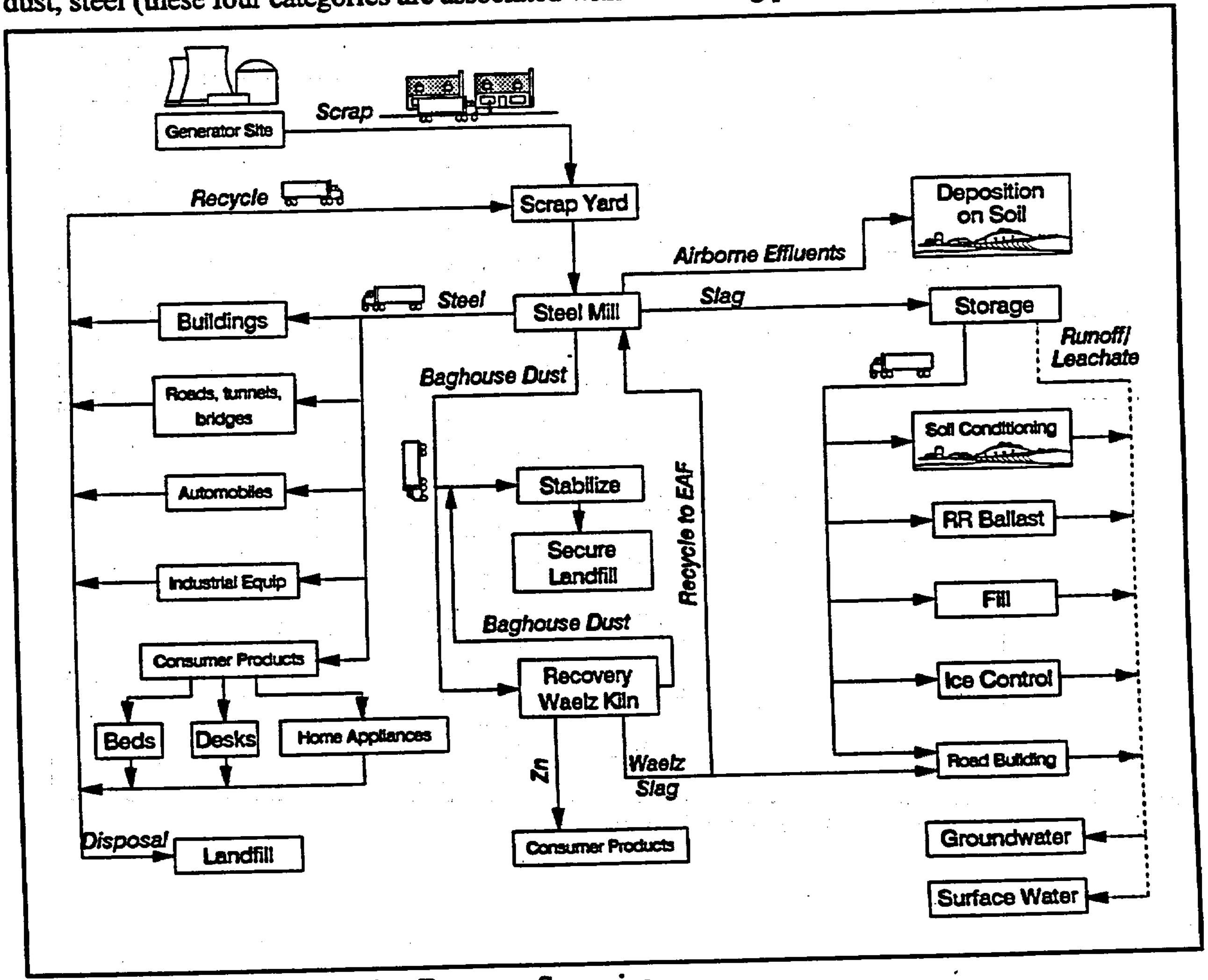

Figure 9-1: Potential Collective Exposure Scenarios

Figure 9-2 shows the approach that was used to develop the total normalized collective doses and risks. As shown on the right side of Figure 9-2, for each of the four resulting media, one or more initial re-use and/or final disposition scenarios were postulated. Normalized collective doses were determined for each of these scenarios on the basis of unit activity. These are referred to as the unweighted scenario normalized collective doses. Next, the unweighted scenario normalized collective doses for each media type were combined with the expected usage factor for each scenario to arrive at media specific normalized collective doses. The calculation of these normalized collective doses is presented in Sections 9.2 through 9.5. Finally, as presented in Section 9.6, the media specific normalized collective doses were combined with the radionuclide specific partitioning factors (a few radionuclides partition to more than one of the media shown on Figure 9-2; see Table 6-3) and the transportation normalized collective doses to determine the radionuclide specific total normalized collective doses and risks. (See Table 9-15.)

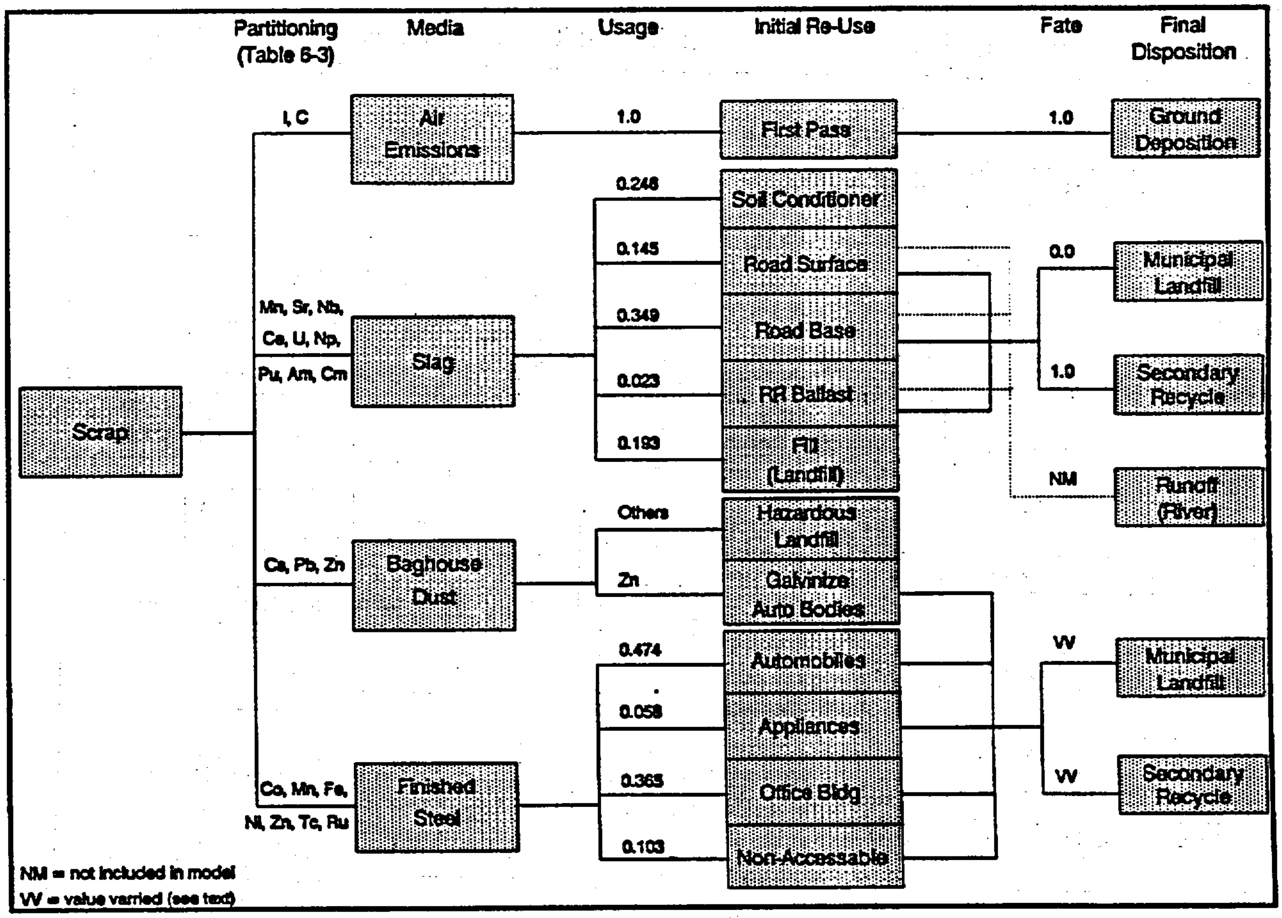

Figure 9-2: Collective Impact Calculational Approach

The discussion in the following sections focuses on doses and dose models; however, the potential cancer risks from the recycling of scrap metal from nuclear facilities were also determined. For external exposures, cancer risks were determined by multiplying the calculated dose by 7.6×10^{-4} total cancers per rem and 5.1×10^{-4} fatal cancers per rem dose-to-risk factors, as recommended by EPA 94. For internal exposure, cancer risks were determined using the same models presented below for calculating doses, except that slope factors from EPA 94a were used in place of dose conversion factors.

9.1 TRANSPORTATION

Most of the scrap material potentially recycled from nuclear facilities—either the scrap metal itself or the products and by-products of melting—would be transported by truck or by rail. The method used to estimate the radiological impacts on the population residing along the transportation routes from shipments of contaminated material is the same as that used by RADTRAN and IMPACTS-BRC (O'Neal 90). The following equation was used to calculate the collective dose to the general population from external exposure to a single shipment of a material:

$$\Delta_{it} = \frac{4\cdot10^{-3}\, P\ L\ K_i}{V}\int_{x_m}^{\infty}dx\int_{0}^{\infty}dy\ \frac{e^{-\mu_a r}B(\mu_a,r)}{x^2+y^2} \tag{1}$$

Δ_{it} = collective dose from exposure to radionuclide i during shipment t (person-rem)
P = population density (m^{-2})
L = length of trip (mi)
K_i = source strength of radionuclide i (mrem-m^2/hr)
V = speed of truck
= 40 mi/hr
x_m = distance to nearest receptor
= 30 ft (9.14 m)
μ_a = linear attenuation coefficient of air (m^{-1})
x = distance from roadway to receptor
y = distance along roadway from truck to receptor

$B(\mu_a r)$, the Berger buildup factor for air, is represented by the following expression:

$$B(\mu_a,r) = 1 + a_{aE}\,\mu_a r\, e^{b_{aE}\mu_a r} \tag{2}$$

a_{aE} = first Berger buildup coefficient for gamma radiation of energy E in air
b_{aE} = second Berger buildup coefficient for gamma radiation of energy E in air

The linear attenuation coefficients in air are calculated from the mass attenuation coefficients for air presented in the *Radiological Health Handbook.* The buildup coefficients were taken from Table A4.9 of the *Principles of Radiation Shielding*, A. B. Chilton, *et al.*, 1984.

The factor of $4 \cdot 10^{-3}$ accounts for integrating the radiation field of the truck in both directions (both approaching and receding from the receptor point) and over the population on both sides of the road, as well as converting from millirem to rem.

The source strength, K_i, is the dose rate at a point one meter from an equivalent point source at the center of the truck. The source strength was evaluated by calculating the dose rate at a point 100 meters *outside* the truck without any attenuation by the intervening air and extrapolating it to a point one meter from the center.

In this analysis, six transportation segments were examined, as shown in Table 9-1. Because most scrap generators, scrap yards, steel mills, etc. are located relatively close together, 100 miles was selected as a representative maximum transport distance. It was assumed that finished products could be shipped anywhere in the country; therefore, a longer transport distance of 1000 miles was assumed. For all transportation segments (as well as for all collective impact analyses in this chapter) a representative population density of 100 people/km^2 was assumed.

Table 9-1: Major Transportation Pathway Assumptions

Material	Source	Destination	Distance	Population Density
Scrap	Generator Site	Scrap Yard	100 mi	100 people/km^2
Scrap	Scrap Yard	Steel Mill	100 mi	100 people/km^2
Steel	Steel Mill	User	1000 mi	100 people/km^2
Steel	User (Recycle)	Scrap Yard	100 mi	100 people/km^2
Baghouse Dust	Steel Mill	Processor	100 mi	100 people/km^2
Slag	Steel Mill	User	100 mi	100 people/km^2

Normalized collective doses from the transportation of scrap metals recycled from nuclear facilities and their products and by-products are shown in Table 9-2.

Table 9-2: Unweighted Transportation Doses
(person-rem per Ci—transported)

Nuclide	Scrap	Steel	Slag	Dust
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	2.42E-03	1.77E-02	3.50E-04	6.91E-03
Fe-55	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Co-60	7.68E-03	5.62E-02	1.07E-03	2.14E-02
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	1.75E-03	1.28E-02	2.49E-04	1.25E-03
Sr-90+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mo-93	6.67E-08	4.98E-07	2.05E-08	6.41E-08
Nb-94	4.50E-03	3.29E-02	6.54E-04	1.29E-02
Tc-99	1.74E-10	1.28E-09	6.00E-11	6.45E-10
Ru-106+D	5.60E-04	4.09E-03	8.34E-05	1.63E-03
Ag-110m	7.98E-03	5.83E-02	1.15E-03	5.71E-03
Sb-125	1.06E-03	7.77E-03	1.64E-04	3.14E-03
I-129	8.19E-07	6.03E-06	2.86E-07	8.93E-07
Cs-134	4.35E-03	3.18E-02	6.38E-04	1.26E-02
Cs-137+D	1.55E-03	1.14E-02	2.29E-04	4.48E-03
Ce-144+D	1.09E-04	7.95E-04	1.71E-05	7.92E-05
Pm-147	2.23E-09	1.64E-08	6.64E-10	8.09E-09
Eu-152	3.16E-03	2.33E-02	4.46E-04	8.63E-03
Pb-210+D	1.81E-07	1.33E-06	6.06E-08	7.38E-07
Ra-226+D	5.04E-03	3.70E-02	7.25E-04	1.42E-02
Ra-228+D	2.60E-03	1.92E-02	3.69E-04	7.17E-03
Ac-227+D	7.09E-04	5.14E-03	1.28E-04	2.28E-03
Th-228+D	4.39E-03	3.24E-02	6.32E-04	1.22E-02
Th-229+D	5.58E-04	4.20E-03	7.96E-05	1.47E-03
Th-230	1.78E-07	1.31E-06	4.84E-08	6.54E-07
Th-232	5.42E-08	3.98E-07	1.71E-08	2.07E-07
Pa-231	5.30E-05	3.88E-04	9.54E-06	1.67E-04
U-234	4.28E-08	3.15E-07	1.33E-08	1.61E-07

Table 9-2: Unweighted Transportation Doses
(person-rem per Ci—transported)

Nuclide	Scrap	Steel	Slag	Dust
U-235+D	1.79E-04	1.32E-03	4.11E-05	6.08E-04
U-238+D	5.10E-05	3.93E-04	5.68E-06	1.11E-04
Np-237+D	3.83E-04	2.81E-03	6.96E-05	1.19E-03
Pu-238	8.67E-09	6.44E-08	2.83E-09	8.69E-09
Pu-239	3.24E-08	2.38E-07	1.00E-08	1.18E-07
Pu-240	8.53E-09	6.33E-08	2.80E-09	8.59E-09
Pu-241+D	2.79E-09	2.04E-08	6.49E-10	2.41E-09
Pu-242	7.48E-09	5.55E-08	2.47E-09	7.56E-09
Am-241	2.97E-06	2.19E-05	1.11E-06	1.28E-05
Cm-244	7.56E-09	5.62E-08	2.41E-09	7.45E-09

9.2 AIRBORNE EMISSIONS

Some fraction of each radionuclide contaminating the scrap metal could be released to the atmosphere when the scrap is melted and converted to steel. Although, as shown in Figure 9-2, only iodine (I-129) and carbon (C-14) have airborne emissions as their primary partitioning medium, other radionuclides could be released in airborne emissions as fugitive dust. Therefore, airborne emission normalized collective doses were calculated for all 40 radionuclides being evaluated.

The radiological impacts of these releases were assessed using the EPA's CAP88-PC computer code, and CU-POP (EPA 94b), a model developed for use in EPA's assessment of the collective impacts of radioactively contaminated soils. CAP88-PC was used to calculate the impacts from the first pass of released radionuclides, while the CU-POP model was used to evaluate long-term (*i.e.*, 1,000 year) impacts from the ground deposition of airborne emissions for all radionuclides except C-14 and I-129.

The CAP88-PC computer program is a desk-top computer version of the earlier CAP-88 code, used to perform dose and risk assessments for the purpose of demonstrating compliance with 40 CFR 61.93(a). CAP88-PC embodies the CAP-88 methodology for radiological assessments of both populations and maximally exposed individuals. The program uses a modified Gaussian plume equation to estimate the average dispersion of radionuclides and rates of deposition on the ground. The radionuclide concentrations in produce, leafy vegetables, milk and meat consumed by humans are estimated by coupling the output of the atmospheric transport models with the U.S. NRC Regulatory Guide 1.109 terrestrial food chain models. Assessments are done for a circular grid within a radius of 80 kilometers (50 miles) around the facility. The mathematical models and equations used in CAP88-PC are presented in the CAP88-PC user's guide.

Time integrated collective doses were calculated from atmospheric data that were provided with CAP88-PC for seven sites located around the country, along with a constant population density of 100 people/km^2 and the default CAP88-PC usage factors. The seven sites are: Providence, RI; Harrisburg, PA; Wilmington, DE; Chicago, IL; Los Angeles, CA; Knoxville, TN; and Moline, IL. These sites were selected because of their proximity to nuclear facilities (usually more than one), and because they represent most areas of the country (two each in the Northeast and Mid-West, and one each in the South, Mid-Atlantic, and West).

Table 9-3 presents the results of the CAP88-PC calculation of normalized collective doses due to the initial release of radionuclides into the air. The average collective dose factor from all the sites was used as the basis for this analysis. However, it should be noted that the doses do not vary significantly from one site to another, largely because the population density was kept constant.

Table 9-3: Unweighted Airborne Emission Doses (person-rem per Ci—released)

Nuclide	First Pass (CAP88-PC)			Soil Deposition (CU-POP)		
	Maximum	Minimum	Average	Maximum	Minimum	Average
C-14	2.54E-01	1.21E-01	1.81E-01	NC	NC	6.48E+01
Mn-54	1.04E+00	7.22E-01	8.64E-01	9.79E-01	9.30E-01	9.57E-01
Fe-55	3.76E-02	2.65E-02	3.16E-02	5.03E-04	4.99E-04	5.01E-04
Co-60	5.59E+00	3.88E+00	4.66E+00	2.36E+01	1.68E+01	1.97E+01
Ni-59	1.29E-02	8.98E-03	1.08E-02	1.90E-01	1.49E-01	1.65E-01
Ni-63	3.26E-02	2.28E-02	2.74E-02	1.19E+00	7.81E-01	9.43E-01
Zn-65	3.25E+00	2.31E+00	2.73E+00	3.63E+00	3.29E+00	3.37E+00
Sr-90+D	5.82E+00	4.10E+00	4.89E+00	4.84E+02	1.67E+02	2.51E+02
Nb-94	7.95E+00	5.54E+00	6.65E+00	1.46E+03	3.46E+00	3.14E+02
Mo-93	7.80E-03	5.42E-03	6.48E-03	2.27E+01	4.61E-01	5.41E+00

Table 9-3: Unweighted Airborne Emission Doses (person-rem per Ci—released)

Nuclide	First Pass (CAP88-PC)			Soil Deposition (CU-POP)		
	Maximum	Minimum	Average	Maximum	Minimum	Average
Tc-99	2.23E-01	1.57E-01	1.87E-01	4.27E+02	2.58E-01	2.34E+01
Ru-106+D	2.89E+00	1.96E+00	2.41E+00	5.78E-01	5.03E-01	5.28E-01
Ag-110m	4.53E+00	3.16E+00	3.78E+00	3.56E+00	3.17E+00	3.28E+00
Sb-125	7.23E-01	5.02E-01	6.02E-01	8.64E-02	7.80E-02	8.02E-02
I-129	2.63E+01	2.00E+01	2.33E+01	NC	NC	1.14E+02
Cs-134	8.23E+00	5.82E+00	6.90E+00	5.78E+00	5.67E+00	5.73E+00
Cs-137+D	4.56E+00	3.25E+00	3.84E+00	3.74E+01	3.03E+01	3.38E+01
Ce-144+D	2.25E+00	1.52E+00	1.88E+00	1.47E-01	1.46E-01	1.46E-01
Pm-147	2.11E-01	1.42E-01	1.76E-01	4.01E-04	3.96E-04	3.98E-04
Eu-152	2.72E+00	1.86E+00	2.26E+00	1.89E+01	1.61E+01	1.75E+01
Pb-210+D	4.80E+02	2.71E+02	3.66E+02	4.93E+02	4.29E+02	4.50E+02
Ra-226+D	1.14E+02	6.62E+01	8.80E+01	1.26E+04	8.24E+03	1.03E+04
Ra-228+D	4.99E+01	2.82E+01	3.81E+01	2.16E+01	2.10E+01	2.13E+01
Ac-227+D	3.24E+03	2.13E+03	2.67E+03	8.04E+01	6.84E+01	7.44E+01
Th-228+D	1.19E+03	7.90E+02	9.87E+02	3.55E+00	3.55E+00	3.55E+00
Th-229+D	3.29E+03	2.20E+03	2.74E+03	6.63E+02	4.95E+02	5.80E+02
Th-230	1.16E+03	7.74E+02	9.66E+02	3.25E+03	2.36E+03	2.80E+03
Th-232	1.67E+03	1.11E+03	1.39E+03	2.92E+03	2.16E+03	2.54E+03
Pa-231	2.34E+03	1.56E+03	1.95E+03	5.31E+04	1.72E+02	4.11E+03
U-234	6.34E+02	4.23E+02	5.27E+02	1.75E+03	2.52E+01	6.64E+02
U-235+D	5.88E+02	3.92E+02	4.89E+02	1.74E+03	2.39E+01	6.43E+02
U-238+D	5.64E+02	3.77E+02	4.70E+02	1.61E+03	2.20E+01	6.16E+02
Np-237+D	2.34E+03	1.56E+03	1.95E+03	3.19E+04	1.30E+02	1.81E+04
Pu-238	1.55E+03	1.03E+03	1.29E+03	1.26E+01	9.30E+00	1.09E+01
Pu-239	1.67E+03	1.11E+03	1.39E+03	1.08E+02	4.22E+01	6.85E+01
Pu-240	1.67E+03	1.11E+03	1.39E+03	1.04E+02	4.11E+01	6.62E+01
Pu-241+D	2.62E+03	1.75E+03	2.17E+03	3.14E-01	2.91E-01	3.03E-01
Pu-242	1.59E+03	1.06E+03	1.32E+03	1.04E+02	4.07E+01	6.61E+01
Am-241	2.59E+03	1.73E+03	2.15E+03	1.40E+02	1.19E+02	1.30E+02
Cm-244	1.37E+03	9.13E+02	1.14E+03	1.30E+00	1.29E+00	1.29E+00

A brief mathematical description of CU-POP is found in Exhibit 9-A. A more detailed description is found in Sections 2.2.5 and E.1 of the TSD for clean up radiologically contaminated soil (EPA 94b). In EPA 94b, a Generic Site and 27 Reference Sites were defined and evaluated. As with the sites selected for the CAP88-PC analysis, the 28 sites evaluated with CU-POP represent most areas of the country. Also, a constant population density of 100 people/km^2 and the EPA 94b usage factors were used. Again, Table 9-3 presents the results of the CU-POP calculation of normalized collective doses due to the ground deposition of radionuclides released into the air. The average collective dose from all 28 sites is used as the basis for the collective dose factor. The total normalized collective doses from the air emission of radionuclides is the sum of the initial release and the subsequent ground deposition normalized collective doses.

For C-14, a global normalized dose of 399 person-rem/Ci over 10,000 years was taken from Section 9.2.6 of NRC 95 and adjusted to the 1,000 year evaluation period being used in this analysis—giving a value of 64.8 person-rem/Ci. For I-129, a global normalized dose of 3,800 rem/Ci to the thyroid over a 1,000 year period is reported in Section 4.8.2 of NCRP 83. This value was converted to an effective dose equivalent by multiplying by 0.03—giving a value of 114 person-rem/Ci.

9.3 SLAG

Slag—material that is a by-product of steel production—accumulates large fractions of many of the radionuclides considered in the present analysis. Consequently, exposure to the slag will account for most of the collective impacts of these nuclides. Slag is usually sold by the steel manufacturer to slag processing companies that in turn re-sell it for a wide range of various uses, including road building, fill material, railroad ballast, soil conditioning, ice control, etc. (see Figure 9-1). The following sub-sections develop exposure scenarios for most of these slag uses, and unweighted normalized collective doses are calculated.

Alternatively, if there is no slag market, the steel manufacturer will stockpile the slag on-site until the slag market returns. In this case, collective exposures during the operation of the steel plant would result from the leaching of contamination from the slag and the off-site consumption of ground water. If a slag pile remains following the closure of the steel plant, collective exposures could result from people living or farming on the abandoned slag piles. Since the current slag market is robust and virtually all of the slag generated is sold for re-use, unweighted normalized collective doses have not been calculated for this stockpiling scenario.

9.3.1 Road-Building

A significant portion of the slag is currently used in road construction, as described below. However, it should be pointed out that in the future this may not be the case. There are problems with expansion from hydration of the slags which causes alligator cracking of surfaces and buckling of substructures and significantly reduces the road's life span. Also, there are some locations (e.g., Maryland) where leachates from the slag have a very high pH and could be considered to be too hazardous to use. Nonetheless, since this analysis is based on current practices, the use of slag in road building was evaluated. (See Chapter 10 for the effect of eliminating this use of slag.)

Road Base

Slag from electric arc furnaces is currently used as the under-layment in road construction. The information in this paragraph is based on a survey of the Federal and seven state departments of transportation. The design life of highways is normally 35-40 years, although primary state highways can be left in place for closer to 60 years. However, highways are seldom simply abandoned at the end of their design life. They are repaired, resurfaced, and/or enlarged and used indefinitely. Furthermore, most states try to recycle all materials used in road building—concrete is crushed and used as aggregate in highway bases, while old asphalt is mixed with new in proportions as high as 50-60%. Materials not used are stockpiled, and if still not used, are given to lower levels of government for use on local roads.

Therefore, it was assumed that slag used in road base construction would remain within the road base for the entire 1,000 year evaluation period. During this time, strong γ-emitting nuclides in this slag expose the motoring public to external radiation.

Assessment of External Exposure

The external exposures of members of the general population traveling on roads built with slag that is a by-product of recycling scrap metal from nuclear facilities are assessed in the following manner:

$$\Delta_{idx}(x) = \frac{10^3 F_{ix}(x)\, f_l\, F_o}{m_d} \left(\frac{1 - e^{-\lambda_i t_r}}{\lambda_i} \right) \qquad (3)$$

$\Delta_{idx}(x)$ = normalized cumulative collective dose to occupants of vehicles from external exposure to nuclide *i* in the road at distance x (person-rem per Ci in scrap)
$F_{ix}(x)$ = road base dose conversion factor (mrem/hr per pCi/g)
f_l = fraction of radionuclide in scrap which partitions to slag
F_o = roadway occupancy factor (see Equation 4)
= 20.56 ($\mathrm{hr \cdot m^{-1} \cdot y^{-1}}$)
m_d = mass of slag in road per unit length (g/m)
t_r = evaluation time
= 1,000 y

The roadway occupancy factor is calculated from the total highway travel in the U.S., as reported by the Federal Highway Administration; the total length of U.S. highways, measured along the centerline of each road (referred to as "racetrack miles"); and an assumed average speed of passenger vehicles.

$$F_o = \frac{D_p}{L_r v} \qquad (4)$$

D_p = total highway travel in U.S.
= 3.87653×10^{12} person-mi/y
L_r = total length of U.S. highways
= 3,904,721 "racetrack" miles
v = Average speed of passenger vehicles
= 30 mph x 1,609 m/mi
= 48,280 m/hr

The road base dose conversion factor [$F_{ix}(x)$] was calculated with the MicroShield™ computer code, and includes the shielding effect of the road pavement. The normalized collective doses from the use of slag as road base are presented in Table 9-4.

Table 9-4: Slag Pathway Normalized Collective Doses (person-rem per Ci—in pathway)

Nuclide	Soil	Concrete	Road Base	RR Ballast	River
C-14	0.00e+00	3.31e-03	0.00e+00	2.54e-03	0.00e+00
Mn-54	9.57e-01	1.65e+00	4.40e-02	1.27e+00	1.01e+00
Fe-55	5.01e-04	0.00e+00	0.00e+00	0.00e+00	1.59e-01
Co-60	1.97e+01	3.21e+01	1.61e+00	2.46e+01	7.11e+00
Ni-59	1.65e-01	0.00e+00	0.00e+00	0.00e+00	9.28e-02
Ni-63	9.43e-01	0.00e+00	0.00e+00	0.00e+00	3.94e-01
Zn-65	3.37e+00	9.32e-01	3.94e-02	7.15e-01	1.54e+01
Sr-90+D	2.51e+02	0.00e+00	0.00e+00	0.00e+00	1.78e+02
Nb-94	3.14e+02	2.48e+03	6.08e+01	1.90e+03	7.39e+01
Mo-93	5.41e+00	1.39e-01	3.78e-24	1.07e-01	1.31e+00
Tc-99	2.34e+01	3.28e-02	8.74e-10	2.51e-02	3.66e+00
Ru-106+D	5.28e-01	5.01e-01	8.84e-03	3.85e-01	6.21e+00
Ag-110m	3.28e+00	4.41e+00	1.46e-01	3.38e+00	4.36e+00
Sb-125	8.02e-02	1.06e-01	1.21e-03	8.16e-02	8.13e-01
I-129	1.41e+03	3.37e+00	3.50e-20	2.58e+00	1.64e+02
Cs-134	5.73e+00	7.32e+00	1.55e-01	5.62e+00	5.25e+01
Cs-137+D	3.38e+01	3.85e+01	6.68e-01	2.95e+01	4.11e+01
Ce-144+D	1.46e-01	9.45e-02	4.38e-03	7.24e-02	8.21e+00
Pm-147	3.98e-04	4.95e-05	6.01e-10	3.80e-05	2.44e-01
Eu-152	1.75e+01	3.59e+01	1.34e+00	2.75e+01	1.55e+00
Pb-210+D	4.50e+02	5.11e-02	1.09e-05	3.92e-02	2.05e+03
Ra-226+D	1.03e+04	2.36e+03	1.13e+02	1.81e+03	3.81e+02
Ra-228+D	2.13e+01	3.48e+01	2.17e+00	2.67e+01	3.51e+02
Ac-227+D	7.44e+01	1.65e+01	9.25e-02	1.27e+01	3.43e+03
Th-228+D	3.55e+00	7.31e+00	5.83e-01	5.61e+00	1.88e+02
Th-229+D	5.80e+02	3.98e+02	5.81e+00	3.05e+02	1.04e+03
Th-230	2.80e+03	5.48e+02	2.61e+01	4.20e+02	1.45e+02
Th-232	2.54e+03	4.17e+03	2.60e+02	3.20e+03	7.27e+02
Pa-231	4.11e+03	4.91e+01	9.99e-02	3.77e+01	3.33e+03
U-234	6.64e+02	1.81e+00	8.14e-02	1.39e+00	7.89e+01

Table 9-4: Slag Pathway Normalized Collective Doses (person-rem per Ci—in pathway)

Nuclide	Soil	Concrete	Road Base	RR Ballast	River
U-235+D	6.43e+02	1.97e+02	8.13e-02	1.51e+02	6.35e+01
U-238+D	6.16e+02	4.05e+01	6.97e-01	3.11e+01	6.37e+01
Np-237+D	1.81e+04	2.85e+02	8.20e-01	2.19e+02	1.80e+03
Pu-238	1.09e+01	4.98e-03	7.54e-14	3.82e-03	7.22e+02
Pu-239	6.85e+01	7.57e-02	1.28e-06	5.80e-02	8.08e+02
Pu-240	6.62e+01	3.63e-02	3.01e-13	2.78e-02	8.07e+02
Pu-241+D	3.03e-01	1.02e-04	4.07e-08	7.85e-05	1.54e+01
Pu-242	6.61e+01	3.34e-02	1.25e-12	2.56e-02	7.68e+02
Am-241	1.30e+02	5.66e+00	4.56e-09	4.34e+00	9.42e+02
Cm-244	1.29e+00	8.57e-04	2.76e-14	6.57e-04	4.66e+02

Leachate

In addition to the direct exposure pathway, some nuclides could leach from the slag as a result of infiltration of rainwater through pores or cracks in the pavement or laterally from overflowing ditches. Some of this leachate would percolate through the soil to an underlying aquifer that is a potential source of drinking water. Other fractions may drain into streams and rivers, while still others may inundate fields and contaminate agricultural soils. Although the present analysis does not attempt to quantify the amount of radioactivity which is expected to travel these routes, normalized collective doses from radionuclide release into a river system and soil contaminated with radionuclides are presented for comparison. The impact of radionuclides leaching into rivers and streams was assessed using EPA methodologies described in *Environmental Pathway Models for Estimating Population Health Effects from Disposal of High-level Radioactive Waste in Geologic Repositories*, EPA 520/5-85-026, J. M. Smith *et al.*, 1985. Normalized collective doses from radionuclides which leach from the road base and reach a river are presented in Table 9-4.

Asphaltic Concrete Aggregate

Thirteen percent of the slag is used as asphaltic concrete aggregate, which is used primarily for paving roads. The slag constitutes 80% of the pavement. As stated above, at the end of its design life, most material used in road construction is recycled into new roads. Therefore, as with slag used in road base construction, all slag used for road paving was assumed to be in place for the entire 1,000 year evaluation period. The equations used to evaluate the collective dose from slag used as asphaltic concrete aggregate are the same as those used to evaluate the road base collective dose, except that the value of the dose conversion factor [$F_{ix}(x)$] differs. The dose conversion factor used for asphaltic concrete aggregate are 80% of those given in Federal Guidance Report No. 12 (EPA 93), to account for the slag component of the pavement. The normalized collective doses for radionuclides contained in slag used as asphaltic concrete aggregate are shown in Table 9-4 (Concrete).

9.3.2 Fill

Slag is often used as fill. Typical fill applications include parking lots and road shoulders. Slag is not recommend for fill applications under buildings because of stability concerns. For this analysis it was assumed that slag used as fill would be covered with a layer of soil and/or concrete, for example, if it were used in the construction of a parking lot. The potential exposures from the soil-covered slag were assessed by modeling the fill as a municipal landfill (i.e., buried contamination without extraordinary protection features). With these assumptions, the primary exposure pathway from slag used as fill is ground water contamination. Alternatively, if it had been assumed that the slag used as fill was uncovered, then it could be assumed that people live on and grow their food in the slag, similar to the exposure pathways developed in Section 9.3.4 for slag used as soil conditioner.

The impacts of slag used as fill (and modeled as a municipal landfill) were assessed with the Multimedia Environmental Pollutant Assessment System (MEPAS) (Buck 95) which is a physics-based risk computation code that integrates source-term, transport, and exposure models. Developed by Pacific Northwest Laboratory for screening and ranking of environmental problems, MEPAS is designed for site-specific assessments using readily available information to estimate potential health impacts. This system has wide applicability to a range of environmental problems using air, ground water, surface water, overland, and exposure models.

Whenever available and appropriate, EPA guidance and models were used to facilitate compatibility and acceptance. The methodology is illustrated in Figure 9-3, below[1]

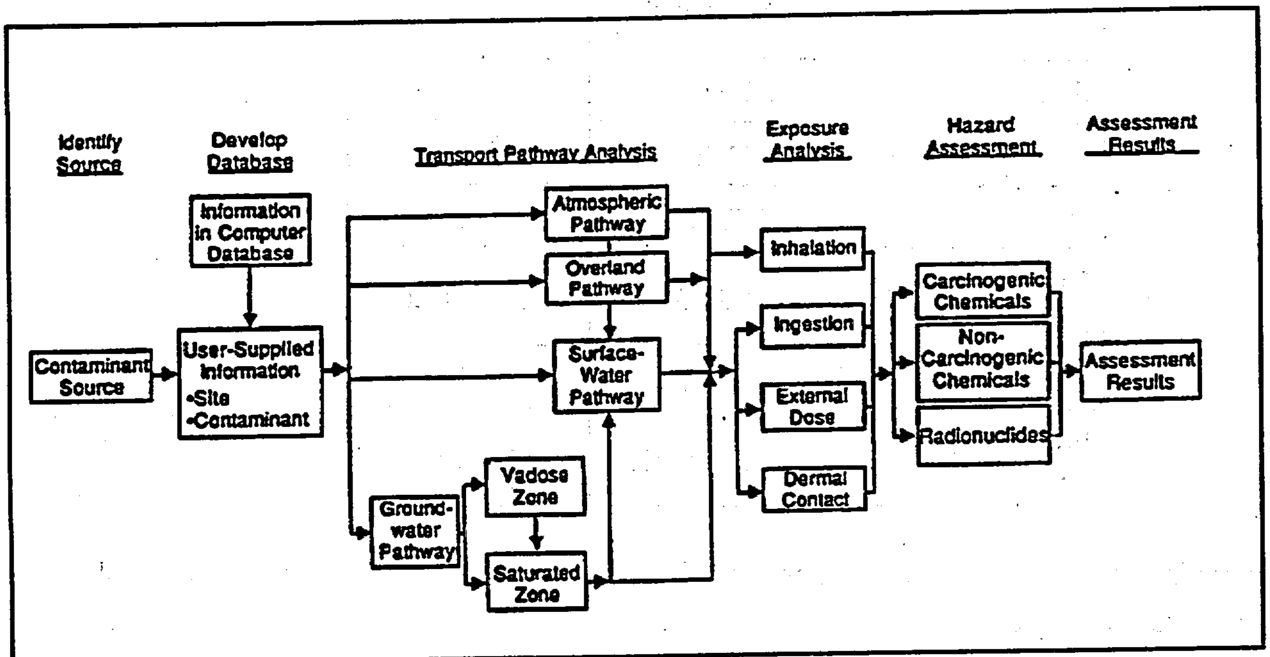

Figure 9-3: Simplified Flow Diagram of the MEPAS Methodology

The exposure scenario assumed for the normalized collective dose model is that ground water would come into contact with the disposed material and that this ground water would in turn be consumed by the down-gradient population. The normalized collective dose model <u>does not</u> include people living directly over the fill who may inadvertently intrude into the fill. The municipal landfill characteristics shown below on Table 9-5 come primarily from NUREG/CR-6147 (Dehmel 94). The information comes from a 1986 EPA survey: Solid Waste Landfill Survey, OMB No. 2050-0061. The survey was a stratified sample of 1,076 respondents of which 1,011 observations were used for analysis. The value used for aquifer thickness is the average value in Newell 90.

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[1] From "Overview of the Multimedia Environmental Pollutant Assessment System (MEPAS)," G. Whelan *et al.*, *Hazardous Waste & Hazardous Materials*, 9:191, 1992

Table 9-5: Typical Landfill Values Assumed

Parameter	Municipal	Hazardous Waste
Size	32.5 acres	18 acres
Depth	11 feet	28 feet
Capacity	3.75E+05 yd^3	8.6E+05 yd^3
Distance to nearest private well	1,850 feet	2,250 feet
Depth to Aquifer	30 ft	49 feet
Population within 1 mile	3,684 people	481 people

The normalized collective doses for slag used as fill are shown in Table 9-6 under the municipal and hazardous waste landfill scenario.

Table 9-6: Landfill Normalized Collective Doses (person-rem per Ci—disposed)

Nuclide	Hazardous	Municipal
C-14	NC	5.75e+01
Mn-54	NC	NC
Fe-55	8.18e-16	1.03e+00
Co-60	0.00e+00	1.14e-17
Ni-59	NC	NC
Ni-63	0.00e+00	6.91e-09
Zn-65	0.00e+00	0.00e+00
Sr-90+D	0.00e+00	5.54e-11
Nb-94	1.15e+03	3.02e+02
Mo-93	2.41e+02	1.29e+01
Tc-99	1.64e-30	1.05e+02
Ru-106+D	0.00e+00	0.00e+00
Ag-110m	NC	NC

Table 9-6: Landfill Normalized Collective Doses (person-rem per Ci—disposed)

Nuclide	Hazardous	Municipal
Sb-125	0.00e+00	2.03e-12
I-129	NC	NC
Cs-134	NC	NC
Cs-137+D	0.00e+00	1.14e-19
Ce-144+D	NC	NC
Pm-147	6.73e-16	3.54e+00
Eu-152	NC	NC
Pb-210+D	0.00e+00	NC
Ra-226+D	0.00e+00	5.57e+00
Ra-228+D	9.14e-06	1.06e-32
Ac-227+D	0.00e+00	0.00e+00
Th-228+D	0.00e+00	4.31e-34
Th-229+D	0.00e+00	NC
Th-230	0.00e+00	2.38e-34
Th-232	0.00e+00	6.25e-34
Pa-231	0.00e+00	1.27e-04
U-234	0.00e+00	5.26e+00
U-235+D	0.00e+00	9.42e+00
U-238+D	0.00e+00	8.86e-02
Np-237+D	6.25e-09	3.62e+03
Pu-238	NC	NC
Pu-239	0.00e+00	9.50e+01
Pu-240	NC	NC
Pu-241+D	0.00e+00	NC
Pu-242	NC	NC
Am-241	0.00e+00	NC
Cm-244	NC	NC

9.3.3 Railroad Ballast

Slag is sometimes used as railroad ballast. The collective impacts from slag used as railroad ballast were calculated in the same manner as the impacts from slag used as a road base, except that the values of the dose conversion factor [$F_{ix}(x)$] and the occupancy factor [F_o] differ. Since the slag used as ballast would be uncovered, the railroad dose conversion factors were taken from Federal Guidance Report No. 12. The railroad occupancy factor of 4.56 $hr \cdot m^{-1} \cdot y^{-1}$ was calculated from 1993 data obtained from the American Public Transit Association. The normalized collective doses from slag used as railroad ballast are shown in Table 9-4.

Because it is exposed to the environment, such slag will be susceptible to leaching. The leachate may either percolate through the underlying soil and potentially reach an aquifer, run off and infiltrate nearby agricultural soils, or drain into rivers and streams. As discussed above, this pathway was not explicitly evaluated in this analysis, however, normalized collective doses from the radionuclides that contaminate agricultural soils and drain into rivers are provided for comparison.

9.3.4 Other Purposes

Other purposes for which slag is used includes soil conditioning, ice control, and miscellaneous uses.

Soil Conditioning

Slag is used to raise the pH of acidic soils—this process is also known as "liming." The resulting population impacts were assessed by means of the CU-POP model. (See Exhibit 9-A for a brief description of the CU-POP model and Section 9.2 for how CU-POP was applied in this analysis to model soil contaminated with radionuclides.) The soil conditioning normalized collective doses shown in Table 9-4 are identical to the average normalized collective doses from the deposition of air emissions shown in Table 9-3.

Ice Control

Slag is spread on ice- or snow-covered roadways to provide traction (see attached memo). After the precipitation has melted, the slag would be removed by street sweepers or washed into storm sewers or roadside ditches. Therefore, after a relatively short time (~4 months), slag used for ice control would be removed from the roadways. The ice control slag use alternative was not evaluated for the following reasons: 1) the breakdown between ice control and soil conditioning is unknown, 2) the breakdown between how much ice control slag ends-up in a waterway and how much end-up on the surrounding land is unknown, 3) the slag spends a short time on the road, and 4) some states do not use slag for ice control (e.g., Michigan, southern states. All slag used for 'Other Purposes' was assumed to be for soil conditioning since much of the slag used for ice control could be washed onto the surrounding land.

9.3.5 Slag Normalized Collective Doses

Steel slag sales (SOL 93, SOL 95) in thousand metric tonnes are listed in Table 9-7.

Table 9-7: Annual Steel Slag Sales (thousand metric tonnes)

Use	1991	1992	1993	1994	1995
Asphaltic concrete aggregate	1085	903	1090	1140 (14.6%)	1040 (14.5%)
Fill	828	1073	905	1320 (16.3%)	1380 (19.3%)
Railroad ballast	186	224	116	160 (2.1%)	168 (2.3%)
Road bases	3238	2400	2600	3170 (40.6%)	2820 (39.4%)
Other[1]	1623	2256	1900	2000 (25.6%)	1760 (24.6%)
Total	6959	6857	6670	7800	7160

1 -Includes ice control, soil conditioning and misc. uses

The media normalized collective doses for slag were calculated by multiplying the five scenario specific normalized collective doses calculated in Sections 9.3.1 through 9.3.4 by the percentage of slag that was sold in 1995 for each re-use scenario and then summing the results:

$$\Delta_s = \sum_n f_n \, \Delta_n \tag{5}$$

Δ_s = unitized collective dose factor from slag (rem/hr per Ci—slag)
n = slag usage scenario: road base, concrete, fill, RR ballast, soil conditioning
f_n = fractional useage of slag in scenairo *n*, see Table 9-7
Δ_n = unitized collective dose factor for scenanio *n* (rem/hr per Ci)

The resulting slag normalized collective doses are shown in Table 9-15, along with the other media specific normalized collective doses and the total normalized collective doses. If, as discussed in Section 9.3.2, it is desired to model slag used as fill as soil conditioner as opposed to a municipal landfill, then the soil conditioner usage fraction can simply be increased from 0.246 to 0.439; the unitized collective dose factors from slag would increase proportionally.

9.4 BAGHOUSE DUST

The volatile emissions and aerosols that evolve from an electric arc furnace during the melting process are condensed and captured by the baghouse. The resulting dust is removed from the mill by tanker trucks—its assumed fate in this analysis is described in this section.

9.4.1 Zinc Recovery

Approximately 86% of the dust is shipped to a processing plant for the extraction of zinc. Exposure pathways will depend on the uses of the extracted metals and the disposal of the residue. For this analysis, it was assumed that the recovered zinc would be used to galvanize automobile shells. A description of the reference automobile shell is given in Section 9.5.1. Normalized collective doses from radionuclides contained within automobile shells are given on Table 9-9.

9.4.2 Disposal in Landfill

The remainder of the dust is disposed of in a "secure" (i.e., hazardous waste) landfill. The collective radiological impacts of this dust will be small, inasmuch as the dust will be largely isolated from the environment. However, this pathway was nonetheless investigated. As with slag used as fill (Section 9.3.2), the collective impacts from baghouse dust disposed of in a hazardous waste landfill were calculated with the MEPAS computer program.

The normalized collective dose model exposure scenario assumed for disposal in a landfill is that ground water would come into contact with the disposed material and that this ground water in turn would be consumed by the down-gradient population. The normalized collective dose model does not include people living directly over the landfill, who may inadvertently intrude into the landfill.

Information from 51 landfill units at 31 facilities was averaged to provide the typical hazardous waste landfill characteristics used, as shown in Table 9-5. Geohydrology data from the 10 landfills that had complete and consistent information were averaged to provide the aquifer characteristics of effective porosity, total porosity, Darcy velocity, and thickness. Sand was used as the aquifer soil type as a conservative assumption. The resulting normalized collective doses are shown in Table 9-6.

9.5 FINISHED STEEL

Steel is used is used to make a wide variety of finished products. It is beyond the scope of the present analysis to attempt to perform an exposure assessment of each product. An estimate of the impacts was made by breaking up the use of carbon steel into a number of general categories and constructing an exposure scenario to represent each category.

Secondary Recycle

Collective impacts from the recycle of scrap from nuclear facilities are being evaluated over a 1,000 year period. It is unrealistic to assume that any commercial product would continue to function for such a long period of time. Additionally, once a commercial product reaches the end of its useful life it can either be disposed of in a municipal landfill or recycled into another commercial product. In order to investigate the sensitivity of the resultant collective dose impacts to whether the steel in commercial products is recycled, four cases were analyzed: 1) single use, then disposal; 2) 50% recycle, 50% disposal; 3) 90% recycle, 10% disposal; and 4) 100% recycle. In order to simplify the calculation, it was assumed that steel is always recycled into the same product (e.g., steel initially recycled into an automobile stays in an automobile for the entire 1,000 year evaluation period). If all commercial products had the same useful life, then this assumption would not be needed. However, since useful lives vary, it is a good approximation.

9.5.1 Automotive

Occupants of automobiles manufactured from scrap metal recycled from nuclear facilities could be exposed to external radiation from the radioactivity in various automotive components. Three auto components were modeled: the engine, the frame, and the shell. The primary assumptions used to model the three components of the automobile are shown in Table 9-8.

Table 9-8: Primary Assumptions Used in the Automobile Model

Assumption	Engine	Frame	Shell
Weight	368 lb	834 lb	945 lb
Dimensions	23" × 23" × 29"	192" × 71" × 0.2"	Top: 192" × 71" × 0.12" Sides: 192" × 35" × 0.12" each
Density	0.66 gm/cc	7.87 gm/cc	7.87 gm/cc
Distance	70 cm	1 ft	1 ft

For the engine, an effective density was calculated by dividing the weight by the volume, while for the frame and shell the effective thicknesses were calculated by dividing the weight by the product of the density and area. The information in Table 9-8 was used to calculate exposure rates with the MicroShield™ computer code. An effective exposure rate from the total automobile was calculated by:

$$\Delta_a = \frac{\sum_n w_n \, \Delta_n}{\sum_n w_n} \qquad (6)$$

Δ_a = unitized collective dose rate from steel used in an automobile (rem/hr per Ci)
n = component of the automobile: engine, frame, shell
w_n = weight of an automobile component (lb), see Table 9-8
Δ_n = unitized collective dose rate from an automobile component (rem/hr per Ci)

Automobile occupancy was estimated at 1,460 person-hr/yr, based on the assumption that two individuals commute one hour each way to and from work every day. The automobile's effective life was assumed to be 7.3 years.

The 1,000 year integrated collective dose from the recycling of steel released from nuclear facilities into automobiles is shown in Table 9-9. Table 9-9 shows only those radionuclides which partition into steel, as indicated in Table 6-3.

Table 9-9: Automobile Doses (person-rem per Ci—in car)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
Finished Steel: Whole Car				
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	1.05E+03	1.06E+03	1.06E+03	1.06E+03
Fe-55	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Co-60	1.12E+04	1.38E+04	1.70E+04	1.81E+04
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	.00E+00	0.00E+00	0.00E+00
Zn-65	5.49E+02	5.49E+02	5.49E+02	5.49E+02
Mo-93	4.27E-01	8.53E-01	4.21E+00	5.29E+01
Tc-99	1.96E-03	3.93E-03	1.96E-02	2.68E-01
Ru-106+D	3.25E+02	3.26E+02	3.27E+02	3.27E+02
Ag-110m	2.72E+03	2.72E+03	2.72E+03	2.72E+03
Sb-125	7.51E+01	7.51E+01	7.51E+01	7.51E+01
Baghouse Dust: Shell Only				
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	1.59E+03	1.59E+03	1.59E+03	1.59E+03
Fe-55	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Co-60	1.68E+04	2.07E+04	2.56E+04	2.72E+04
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	8.25E+02	8.25E+02	8.25E+02	8.25E+02
Mo-93	8.70E-01	1.74E+00	8.58E+00	1.08E+02

Table 9-9: Automobile Doses (person-rem per Ci—in car)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
Tc-99	3.37E-03	6.74E-03	3.37E-02	4.61E-01
Ru-106+D	4.92E+02	4.94E+02	4.95E+02	4.96E+02
Ag-110m	4.10E+03	4.10E+03	4.10E+03	4.10E+03
Sb-125	1.14E+02	1.14E+02	1.14E+02	1.14E+02

9.5.2 Kitchen Appliances

Most major home appliances are located in, or near, the kitchen. Therefore, in order to estimate exposures from steel recycled into home appliances, a representative kitchen was modeled. Seven appliances were assumed to be in, or near, the kitchen: refrigerator, stove, dishwasher, microwave oven, trash compactor, washer and dryer. Furthermore, it was assumed that the kitchen was divided into two areas—a work area where the food is prepared and a dining area where the family gathers to eat, talk, do homework, etc. Major assumptions used in the kitchen model are given in Table 9-10.

Table 9-10: Kitchen Model Major Assumptions

Appliance	Weight (lb)	Dimensions (in) (h × w × d)	Distance (ft)	
			Work Area	Dining Area
Refrigerator	350	69.75 × 35.75 × 26.375	5	15
Stove	166	45.5 × 30 × 28	1	15
Dishwasher	81	34 × 24 × 23.75	5	15
Microwave	70	16.5 × 30 × 14	5	15
Trash Compactor	140	34.25 × 15 × 24.25	5	15
Washer	160	36 × 27 × 25.5	10	20
Dryer	116	36 × 27 × 28.5	10	20

Information concerning the weight and dimensions for most of the appliances listed in Table 9-10 was obtained from the General Electric Answer Center in Louisville, Kentucky. The life expectancy for each appliance was assumed to be 18 years. In selecting the distances at which to calculate the dose rates, a representative kitchen approximately 10' × 20' was assumed. It was further assumed that a person would be working in front of one appliance and would be an approximate equal distance from the others, except for the washer and dryer which would be located in an out-of-the-way location. Likewise, the dining area was assumed to be on the opposite side of the kitchen from the appliances.

An effective thickness for each appliance was determined from its weight and dimensions. Each appliance was modeled as two slabs—representing the front and back panels of each appliance—of this effective thickness and the above assumed height and width. Exposures rates at the above assumed distances were modeled by use of the MicroShield™ computer code.

To estimate kitchen occupancy times, a family of four was assumed. Furthermore, it was assumed that all members of the family eat breakfast in the kitchen seven day a weeks and eat dinner in the kitchen five days a week. Weekday lunch was assumed to be eaten in the kitchen by only one member of the family. It was also assumed that one member of the family would spend one hour each week night doing homework in the kitchen. Based on these assumptions, kitchen occupancy times of 70 person-minutes per day in the work area and 190 person-minutes per day in the dining area were calculated.

The normalized collective doses from exposure to home appliances are shown in Table 9-11. These doses are the weight average of the normalized collective doses for each of the seven appliances being evaluated (similar to equation 6 for the automobile).

Table 9-11: Finished Steel: Kitchen (Seven Appliances) Doses
(person-rem per Ci—in appliance)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	3.14E+02	3.14E+02	3.14E+02	3.14E+02
Fe-55	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Co-60	4.84E+03	5.08E+03	5.29E+03	5.34E+03

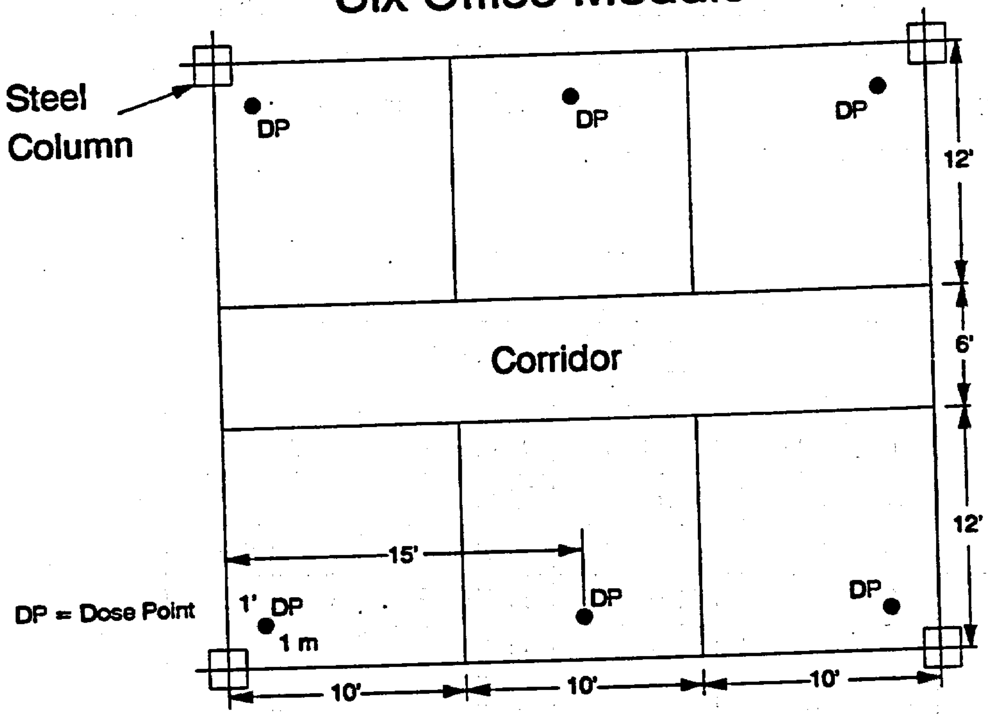
Table 9-11: Finished Steel: Kitchen (Seven Appliances) Doses
(person-rem per Ci—in appliance)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	1.62E+02	1.62E+02	1.62E+02	1.62E+02
Mo-93	9.62E-01	1.92E+00	9.29E+00	4.87E+01
Tc-99	2.14E-03	4.28E-03	2.13E-02	1.20E-01
Ru-106+D	9.78E+01	9.78E+01	9.78E+01	9.78E+01
Ag-110m	8.08E+02	8.08E+02	8.08E+02	8.08E+02
Sb-125	2.26E+01	2.26E+01	2.26E+01	2.26E+01

9.5.3 Office Buildings

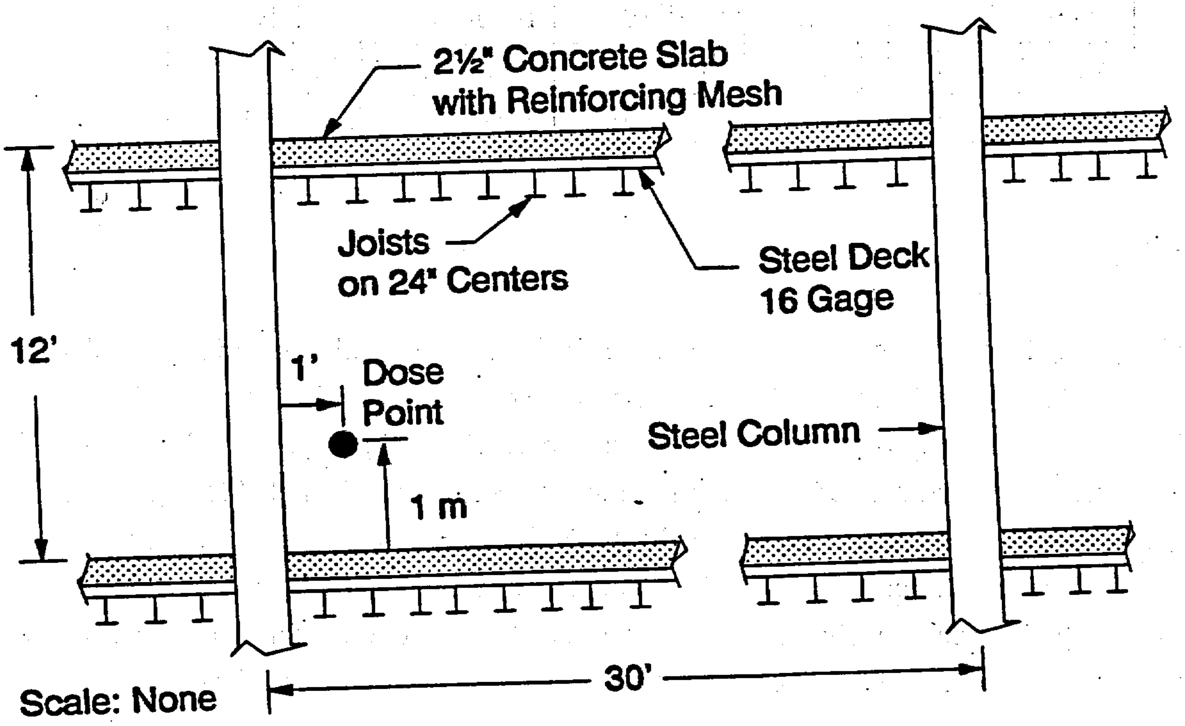
An office building has been selected to represent the exposure from finished steel used in construction projects. The office building was assumed to have a modular construction, with six offices being located within each module. The general layout of a single module and some of the construction details which were assumed are shown on Figure 9-4.

Six Office Module



Scale: None

Office Construction Details



Scale: None

The MicroShield™ computer code was used to calculate the dose rate to the six office workers from each component of the module: columns, floor and ceiling joists, steel wall studs, steel floor and ceiling decking. Each worker was assumed to spend 2,000 hr/yr in the office, and the office module was assumed to have a 50 year service life. The normalized collective doses from steel used in an office building are given in Table 9-12.

Table 9-12: Finished Steel: Office Building Doses
(person-rem per Ci—in office)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	3.78E+02	3.78E+02	3.78E+02	3.78E+02
Fe-55	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Co-60	6.84E+03	6.84E+03	6.85E+03	6.85E+03
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	2.04E+02	2.04E+02	2.04E+02	2.04E+02
Mo-93	3.86E-01	7.64E-01	3.18E+00	7.01E+00
Tc-99	3.32E-03	6.64E-03	2.91E-02	6.63E-02
Ru-106+D	1.12E+02	1.12E+02	1.12E+02	1.12E+02
Ag-110m	9.80E+02	9.80E+02	9.80E+02	9.80E+02
Sb-125	2.50E+01	2.50E+01	2.50E+01	2.50E+01

9.5.4 Cookware

Radioactivity may leach into food from cookware made from scrap metal originating at nuclear facilities, and subsequently be ingested by members of the public. The information used to model this potential pathway is primarily from three sources: SMA 87, KUL 92, and REI 85. SMA 87 presents a national (United Kingdom) average ingestion rate of 0.12±10% mg/day of nickel attributable to stainless steel cookware. KUL 92 gives the results of tests using 5% acetic acid boiled for five minutes in stainless steel cookware. Although not strictly applicable to ordinary cooking, the results can be used to estimate the relative amounts of iron, chromium and nickel that are likely to be leached. The three utensils that most resemble American stainless steel cookware (two were manufactured in the U.S. and one in Brazil) had average concentrations of 0.267 mg Fe, 0.0963 mg Ni, and 0.0387 mg Cr per kilogram of water. From this, it is deduced that the total mass of metal leached is 4.17 times the mass of nickel.

Combining the data from these two references implies that the average total annual ingestion of metal in food cooked in stainless steel cookware is 183±18.3 mg. Alternatively, the metal ingested from food cooked in cast iron cookware can be inferred directly from data in REI 85. From REI 85's concentrations of iron in beef and cabbage cooked in cast iron utensils, an average total ingestion of metal is 1,950±680 mg.

A family of four was assumed to use a 350 gm frying pan. The normalized collective doses from the ingestion of metal in food cooked in a frying pan was integrated over the time it took for all of the metal initially in the frying pan to be consumed (~480 years for stainless steel and ~45 years for cast iron). The resulting normalized collective doses are shown in Table 9-13.

Table 9-13: Finished Steel: Frying Pan Doses
(person-rem per Ci—in pan)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
Cast Iron Frying Pan				
C-14	1.14E+02	2.15E+02	7.53E+02	1.89E+03
Mn-54	9.55E+00	9.55E+00	9.55E+00	9.55E+00
Fe-55	6.53E+00	6.54E+00	6.56E+00	6.56E+00
Co-60	5.25E+02	5.43E+02	5.59E+02	5.63E+02
Ni-59	9.86E+00	1.51E+01	2.62E+01	3.22E+01
Ni-63	2.95E+01	5.00E+01	1.14E+02	1.66E+02
Zn-65	3.91E+01	3.91E+01	3.91E+01	3.91E+01
Mo-93	7.34E+01	1.39E+02	4.81E+02	1.19E+03
Tc-99	7.98E+01	1.51E+02	5.35E+02	1.37E+03
Ru-106+D	1.14E+02	1.14E+02	1.14E+02	1.14E+02
Ag-110m	2.97E+01	2.97E+01	2.97E+01	2.97E+01

Table 9-13: Finished Steel: Frying Pan Doses
(person-rem per Ci—in pan)

Nuclide	Secondary Recycle			
	0%	50%	90%	100%
Sb-125	1.31E+00	1.31E+00	1.31E+00	1.31E+00
Stainless Frying Pan				
C-14	1.10E+01	2.17E+01	1.02E+02	4.57E+02
Mn-54	8.99E-01	8.99E-01	8.99E-01	8.99E-01
Fe-55	6.18E-01	6.20E-01	6.22E-01	6.22E-01
Co-60	5.00E+01	5.19E+01	5.35E+01	5.39E+01
Ni-59	9.48E-01	1.49E+00	2.76E+00	3.50E+00
Ni-63	2.83E+00	5.00E+00	1.28E+01	2.11E+01
Zn-65	3.68E+00	3.68E+00	3.68E+00	3.68E+00
Mo-93	7.07E+00	1.40E+01	6.50E+01	2.83E+02
Tc-99	7.68E+00	1.53E+01	7.30E+01	3.38E+02
Ru-106+D	1.07E+01	1.07E+01	1.07E+01	1.07E+01
Ag-110m	2.80E+00	2.80E+00	2.80E+00	2.80E+00
Sb-125	1.23E-01	1.23E-01	1.23E-01	1.23E-01

9.5.5 Finished Steel Normalized Collective Doses

The American Iron and Steel Institute (AISI) compiles statistics on carbon steel markets based on 21 major market classifications and 31 types of steel products (e.g., cold rolled sheets, reinforcing bars, blooms, slabs, billets, etc.). Information included in the AISI annual report for 1995 (AIS 95) has been collapsed into four categories and is summarized in Table 9-14.

Table 9-14: Distribution of Finished Steel into Commercial Products

Commercial Product	AISI Classifications	Percentage
Automobile	automotive; rail transportation	36.5%
Building (Office)	construction and contractors' products; containers, packaging, shipping material; all other AISI classifications	47.4%

Commercial Product	AISI Classifications	Percentage
Home Appliances (including Cookware)	appliances, utensils, cutlery; other domestic and commercial equipment	5.8%
Non-Accessible	oil and gas industry; electrical equipment	10.3%

"Containers, packaging, shipping material" and "all other AISI classifications" were included in the Building (Office) classification because they are a small fraction (~12.5%) of the total carbon steel market and because the normalized collective doses for this classification fall between the normalized collective doses for the other two classifications which were analyzed. "Oil and gas industry" and "electrical equipment" were assumed to be composed primarily of oil derricks, off-shore drilling platforms, and transmission towers, which would not normally be accessible to the general population.

The media normalized collective doses for finished steel were calculated by multiplying the four scenario specific normalized collective doses calculated in Sections 9.5.1 through 9.5.4 by the distribution of steel in each re-use scenario, and then summing the results:

$$\Delta_f = \sum_n f_n \, \Delta_n \tag{7}$$

Δ_f = unitized collective dose factor from finished steel (rem/hr per Ci—steel)
n = steel usage scenario: automobile, appliance, office, frying pan, non-accessible
f_n = distribution of steel in scenario *n*, see Table 9-14
Δ_n = unitized collective dose factor for scenario *n* (rem/hr per Ci)

The resulting finished steel normalized collective doses are shown in Table 9-15, along with the other media specific and the total normalized collective doses.

9.6 TOTAL NORMALIZED COLLECTIVE DOSES AND RISKS

For each radionuclide, the total normalized collective dose and risks were calculated, taking into account the partitioning of the radionuclide to the differing resulting media (i.e., steel, slag, baghouse dust, and air emissions), and transportation associated with each of the media. The total normalized collective doses were calculated by:

$$\Delta_{Ti} = \sum_{m} P^{t}_{mi} \Delta^{t}_{mi} + \sum_{n} P_{ni} \Delta_{ni} \tag{8}$$

Δ_{Ti} = total collective dose factor for radionuclide *i* (person-rem per Ci scrap)
m = transportation media type: scrap, slag, baghouse dust, finished steel
P^{t}_{mi} = transportation partition ratio of radionuclide *i* to media *m*
Δ^{t}_{mi} = radionuclide *i* specific transportation collective dose factor for media *m* (person-rem per Ci)
n = media type: air emission, slag, baghouse dust, finished steel
P_{ni} = partition ratio of radionuclide *i* to media *n*, see Table 6-3
Δ_{ni} = radionuclide *i* specific collective dose factor for media *n* (person-rem per Ci)

For slag, baghouse dust and finished steel, the transportation partition ratios are the same as the media partition ratios; for scrap, the transportation partition ratio is one and for air emissions it is zero. The total transportation normalized collective doses, the media specific normalized collective doses, and the total normalized collective doses are shown in Table 9-15. Also, shown in Table 9-15 are the total normalized collective cancer risks for each radionuclide.

Table 9-15: Total Normalized Collective Dose and Risks (per Ci—in scrap)

Nuclide	Doses (person-rem)						Cancers	
	Trans	Air	Slag	Dust	Steel	Total	Total	Fatal
C-14	0.00E+00	4.74E+01	0.00E+00	0.00E+00	2.12E-02	4.75E+01	2.34E-02	1.62E-02
Mn-54	1.42E-02	2.73E-04	1.67E-01	0.00E+00	3.80E+02	3.80E+02	2.88E-01	1.93E-01
Fe-55	0.00E+00	1.61E-06	3.98E-03	1.64E-17	2.65E-04	4.24E-03	1.57E-07	1.56E-07
Co-60	6.35E-02	1.22E-03	0.00E+00	0.00E+00	1.01E+04	1.01E+04	7.66E+00	5.13E+00
Ni-59	0.00E+00	8.80E-06	0.00E+00	0.00E+00	1.33E-03	1.34E-03	1.18E-06	1.18E-06
Ni-63	0.00E+00	4.86E-05	0.00E+00	0.00E+00	6.84E-03	6.89E-03	6.60E-06	6.58E-06
Zn-65	5.31E-03	2.44E-02	0.00E+00	1.65E+02	6.13E+01	2.26E+02	1.72E-01	1.15E-01
Sr-90+D	0.00E+00	6.40E-02	5.87E+01	3.09E+00	0.00E+00	6.18E+01	2.25E-02	1.78E-02
Nb-94	5.77E-03	8.02E-02	5.35E+02	2.81E+01	0.00E+00	5.63E+02	3.81E-01	2.54E-01
Mo-93	5.60E-07	2.71E-04	0.00E+00	0.00E+00	2.53E+01	2.53E+01	1.92E-02	1.29E-02
Tc-99	1.45E-09	1.18E-03	0.00E+00	0.00E+00	1.91E-01	1.92E-01	1.58E-04	1.24E-04
Ru-106+D	4.63E-03	1.47E-04	0.00E+00	0.00E+00	1.76E+02	1.76E+02	1.34E-01	9.01E-02
Ag-110m	6.58E-02	3.53E-04	0.00E+00	0.00E+00	1.49E+03	1.49E+03	1.13E+00	7.61E-01
Sb-125	8.78E-03	3.41E-05	0.00E+00	0.00E+00	4.02E+01	4.02E+01	3.05E-02	2.05E-02
I-129	8.19E-07	1.54E+03	0.00E+00	0.00E+00	0.00E+00	1.54E+03	1.01E+00	1.04E-01
Cs-134	1.64E-02	6.00E-02	1.33E-01	0.00E+00	0.00E+00	2.09E-01	1.29E-04	8.68E-05
Cs-137+D	5.82E-03	1.79E-01	7.42E-01	0.00E+00	0.00E+00	9.27E-01	6.39E-04	4.27E-04
Ce-144+D	1.29E-04	5.07E-04	5.04E-02	2.65E-03	0.00E+00	5.36E-02	2.73E-05	1.74E-05
Pm-147	3.27E-09	4.41E-05	6.49E-01	3.42E-02	0.00E+00	6.83E-01	2.05E-06	1.09E-06

Table 9-15: Total Normalized Collective Dose and Risks (per Ci—in scrap)

Nuclide	Doses (person-rem)						Cancers	
	Trans	Air	Slag	Dust	Steel	Total	Total	Fatal
Eu-152	4.02E-03	4.94E-03	1.01E+01	5.34E-01	0.00E+00	1.07E+01	7.77E-03	5.17E-03
Pb-210+D	9.19E-07	4.08E+00	0.00E+00	0.00E+00	0.00E+00	4.08E+00	3.67E-04	2.90E-04
Ra-226+D	6.44E-03	2.60E+00	2.81E+03	1.48E+02	0.00E+00	2.96E+03	7.82E-01	5.49E-01
Ra-228+D	3.31E-03	1.49E-02	1.12E+01	5.88E-01	0.00E+00	1.18E+01	8.11E-03	5.41E-03
Ac-227+D	9.45E-04	6.86E-01	2.00E+01	1.05E+00	0.00E+00	2.17E+01	3.67E-03	2.50E-03
Th-228+D	5.60E-03	2.48E-01	2.18E+00	1.15E-01	0.00E+00	2.55E+00	1.73E-03	1.16E-03
Th-229+D	7.07E-04	8.30E-01	1.99E+02	1.05E+01	0.00E+00	2.11E+02	8.09E-02	5.47E-02
Th-230	2.57E-07	9.42E-01	7.49E+02	3.94E+01	0.00E+00	7.89E+02	2.11E-01	1.53E-01
Th-232	8.08E-08	9.83E-01	1.34E+03	7.03E+01	0.00E+00	1.41E+03	9.18E-01	6.15E-01
Pa-231	7.04E-05	1.52E+00	9.68E+02	5.10E+01	0.00E+00	1.02E+03	2.07E-02	1.46E-02
U-234	6.35E-08	2.98E-01	1.55E+02	8.18E+00	0.00E+00	1.64E+02	2.59E-02	1.60E-02
U-235+D	2.48E-04	2.83E-01	1.82E+02	9.60E+00	0.00E+00	1.92E+02	5.35E-02	3.46E-02
U-238+D	6.19E-05	2.72E-01	1.50E+02	7.92E+00	0.00E+00	1.59E+02	4.02E-02	2.50E-02
Np-237+D	5.09E-04	5.01E+00	4.94E+03	2.60E+02	0.00E+00	5.20E+03	3.41E-01	2.78E-01
Pu-238	1.18E-08	3.25E-01	2.55E+00	1.34E-01	0.00E+00	3.01E+00	2.23E-04	1.97E-04
Pu-239	4.78E-08	3.65E-01	3.34E+01	1.76E+00	0.00E+00	3.56E+01	1.29E-03	1.12E-03
Pu-240	1.16E-08	3.64E-01	1.55E+01	8.15E-01	0.00E+00	1.67E+01	1.24E-03	1.08E-03
Pu-241+D	3.53E-09	5.43E-01	7.08E-02	3.73E-03	0.00E+00	6.17E-01	6.79E-05	5.88E-05
Pu-242	1.02E-08	3.47E-01	1.55E+01	8.13E-01	0.00E+00	1.66E+01	1.23E-03	1.07E-03
Am-241	4.66E-06	5.70E-01	3.13E+01	1.65E+00	0.00E+00	3.35E+01	3.57E-03	2.86E-03
Cm-244	1.02E-08	2.85E-01	3.02E-01	1.59E-02	0.00E+00	6.03E-01	3.37E-05	2.92E-05

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EXHIBIT 9-A

CU-POP: A MODEL FOR ASSESSING THE COLLECTIVE IMPACTS OF RADIOACTIVELY CONTAMINATED SOIL

CU-POP is a simple, generally conservative model developed to estimate the radiological impacts of contaminated soil on both on-site and off-site populations. The model accounts for radioactive decay and the ingrowth of progeny and for the transit time of contaminants through the unsaturated zone.

The five specific exposure pathways addressed in CU-POP are: 1) external exposure to penetrating radiation from contaminated soil; 2) inhalation of suspended dust; 3) exposure to indoor radon progeny (when included); 4) ingestion of crops raised on contaminated soil; and 5) ingestion of contaminated ground water. A sensitivity analysis has shown that for the radionuclides and the environmental conditions reflected in the reference sites, other pathways (such as soil ingestion and irrigation) are not important contributors to the radiological impacts.

The first three pathways affect only the population residing on-site. The radiation exposure of any individual at site *i* is assumed to be proportional to the average concentration of the *j*-th radionuclide on the portion of the site where he or she resides. The estimated cumulative, collective radiological health impact on the population (e.g., number of radiogenic cancers induced) attributable to the *k*-th pathway is the sum of the impacts on these individuals, integrated over space and over the time of site occupation—e.g., from $t=0$ to $t = 1000$ years. This contribution to the population impact, R_{ijkt}, can be expressed as

$$R_{ijkt} = N_i \int_0^t \kappa_{ijk} \left[\int_{S_i} C_{ij}(r,t')\, \rho\, \lambda_i\, dS \right] dt' \qquad (1)$$

where $C_{ij}(r,t')$ is the concentration of nuclide *j* at position *r* and time t', λ_i and ρ are the (constant) thickness and density of the contaminated soil layer and S_i its initial area, N_i is the population density (assumed to be uniform and constant), and κ_{ijk} is the appropriate site-, nuclide-, and pathway-specific constant of proportionality for the population impact under consideration. The quantity inside the square brackets represents the total activity of nuclide *j* at time t', and is independent of its spatial distribution. $C_{ij}(r,t)$ depends on $C_{ij}(r,0)$, the concentration at $t = 0$ (the time of site occupancy) and, in cases where nuclide j is a member of a radioactive decay series and $j\text{-}1, j\text{-}2$, *etc.*, refer to its progenitors, on $C_{i,j-1}(r,0)$, $C_{i,j-2}(r,0)$,..., as well.

CU-POP assumes that all food raised on the site is consumed, either on- or off-site, and that a fixed fraction (0.5%) of the activity that reaches the aquifer under the site is also consumed. The radiological exposures via these two pathways are thus also proportional to the total radioactivity, but are independent of the site population density. They can be modeled by Equation 1, but with $N_i = 1$.

Because the collective impact is a function of the total inventory of all nuclides on the site at $t = 0$, but not on the distribution of that activity over the area of the site, it is possible to simplify Equation 1. Since the time evolution (e.g., leaching, radioactive decay, etc.) of a given nuclide at a given site is independent of position, let us set $C_{ij}(r,t') = C_{ij}(r,0) \cdot c_{ij}(t')$ where, by definition, $c_{ij}(0) = 1$. Then Equation 1 may be separated and summed over all nuclides and pathways as:

$$\begin{aligned} R_{it} &= \sum_j \left[\sum_k n_{ik} K'_{ijk} \int_0^t c_{ij}(t')\,dt' \right] \left[\int_{S_i} C(r,0)_{ij}\, \rho\, \delta_i\, dS \right] \\ &= \sum_j K_{ijt}\, Q_{ij} \end{aligned} \qquad (2)$$

where the index k refers to the five exposure pathways modeled by CU-POP; and $n_{ik} = N_i$ if $1 \leq k \leq 3$, otherwise $n_{ik} = 1$.

K_{ijt} of Equation 2 relates the cumulative, collective health impact of radionuclide j at site i to its total activity there and is calculated with CU-POP. Q_{ij} is the total inventory of the nuclide at that site.

CHAPTER 10

EVALUATION OF UNCERTAINTIES PERTAINING TO SCRAP METAL QUANTITIES, ESTIMATES OF DOSE AND RISK, AND MINIMUM DETECTABLE CONCENTRATIONS

10.1 INTRODUCTION

The option to recycle large quantities of scrap metal from the decommissioning of DOE facilities and commercial nuclear power plants may result in small radiation exposures to selected individuals, such as workers, and to members of the general public. It is anticipated that all releases of scrap metal for recycling will require a demonstration that residual contamination levels meet applicable regulatory guidelines.

At present, however, regulatory guidelines aimed specifically at scrap metal recycling from nuclear facilities do not exist. As a result, it is not possible at this time to make rigid estimates of future exposure and risks to individuals and to population groups. In acknowledgment of this limitation, EPA calculated radionuclide-specific dose estimates for individuals are based on a unit concentration (i.e., mrem per pCi/g of residual contamination in scrap metal) and for population groups on a Curie quantity throughput of scrap metal (i.e., person-rem per Ci of radioactivity in scrap metal). Exposure estimates of this type are referred to as "normalized."

Chapters 7 and 9 provided detailed estimates of normalized individual and collective doses, respectively, for a variety of modeled exposure scenarios. Scenario selection for these modeled dose estimates are considered reasonable but will likely yield high end doses. The relationship between a starting contamination level in scrap metal and the estimated dose(s) to an individual or group of individuals is uncertain. Modeled dose estimates require values to be assigned to a large number of variables referred to as model parameters. In some instances, values of model parameters may be unknown. In other instances, the values of model parameters, even if known, are highly variable.

In the past, models that employed single conservative values for each parameter resulted in overestimated doses. Such conservative models were employed as screening tools but otherwise had limited value in risk and regulatory impact analyses.

Emphasis is now being placed on removing conservative assumptions and incorporating "realism" into model predictions. Accordingly, dose calculations and risk assessments are tailored, whenever possible, to actual locations or conditions of exposed persons. They incorporate site-specific meteorological data, food production/consumption information, and demographic information as much as possible. Many critical model parameters cannot, however, be determined on a site-specific basis. This is especially true for a model that attempts to predict exposures to future population groups that have yet to be identified.

Attempts at improving the realism of model predictions by removing conservative assumptions, despite uncertainties, increase the probability of underestimating human dose. When examining the results of such evaluations, it is reasonable to question their accuracy and to ask whether actual doses might exceed regulatory standards. To address this issue, uncertainty analysis is conducted. Today, uncertainty analysis comprises an integral part of the Agency's scientific and regulatory analyses.

EPA has long recognized the usefulness of uncertainty analysis and the need to provide ranges of estimates rather than point estimates. According to EPA guidance (EPA 95), a complete uncertainty analysis must address both uncertainty and the variability of individual parameters. Uncertainty is concerned with gaps in data or parameter information that is incomplete; parameter variation is concerned with actual variations in values for the parameters.

Deterministic Versus Stochastic Models

When a model employs single values for individual parameters, it is referred to as deterministic; when individual parameters are defined by ranges or distributions of values, the model is probabilistic (or stochastic). Figure 10-1 compares the simple operation of a deterministic model with that of a more complex probabilistic model. When using the deterministic model, one simply chooses values of x and y, and the model calculates a single output value of z. In a probabilistic model, the user specifies the distribution of each variable and the model then "samples" the distribution of x and y to calculate a value of z. When this calculation is repeated many times, a distribution of the output values of z is generated; this is known as a Monte Carlo simulation. In this report, Monte Carlo analyses were not performed due to the unavailability of information regarding the distribution of the values of the calculational parameters. However, information was available on the range of values for key parameters. As such, the sensitivity/

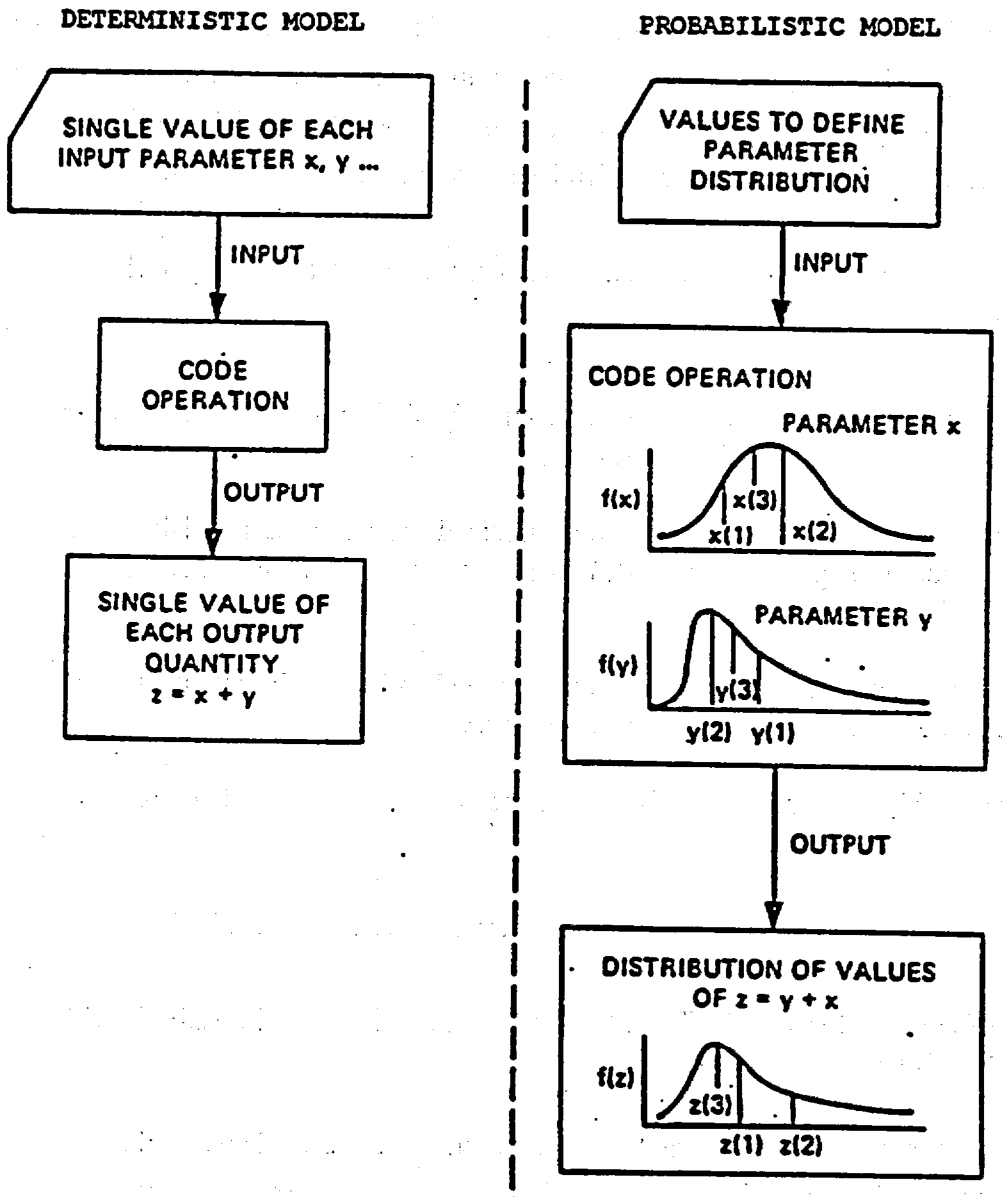

Figure 10-1. Comparison of a Deterministic Model and a Probabilistic Model (from Little 1983)

uncertainty analysis provided in this report evaluated how the results may change over the plausible range of the values for the key calculational parameters. As such, the uncertainty/ sensitivity analyses provided in this report are referred to as "semi-quantitative." Examples of some of these parameters include (1) the partitioning and fate of individual radionuclides during the smelting process, (2) disposition or commercial uses of products and side-streams associated with metal smelting, (3) probable ratios of contaminated scrap to clean scrap that would serve to dilute activity levels in finished products, as well as smelting byproducts, and (4) conditions of exposure for scrap metal workers and end-users of finished products and byproducts.

This chapter summarizes the uncertainties and sensitivities for each of the four major elements defined in the TSD:

(1) Quantities and characteristics of scrap metals from DOE facilities and nuclear power plants that are potentially available for recycling.

(2) Radionuclide-specific estimates of potential normalized annual doses and risks to the reasonably maximally exposed individual (RMEI) associated with the free release of scrap metal from nuclear facilities.

(3) Radionuclide-specific estimates of potential normalized collective doses/risks to the exposed population due to free release of scrap metal.

(4) Minimum detectable concentrations of radionuclide contaminants associated with scrap metal.

An expanded discussion of these uncertainties is provided in Appendix L of the TSD.

10.2 UNCERTAINTIES IN SCRAP METAL SOURCE QUANTITIES AND LEVELS OF CONTAMINATION

Uncertainties pertaining to scrap quantities and contamination levels have widely differing impacts on dose estimates. Uncertainty of scrap metal quantity is <u>not</u> expected to have a significant impact on RMEI doses since these are principally dictated by the radionuclide concentrations (e.g., pCi/g) of scrap at the point of release, and not on the quantity of scrap recycled. In contrast, uncertainty in scrap metal quantities is expected to significantly impact the estimated collective doses since these are proportional to the throughput of scrap metal. For example, for a given radionuclide contamination level, doubling the quantity of recycled scrap metal is expected to double the collective dose.

10.2.1 Scrap Metal from Nuclear Power Plants

The total quantity of scrap metal that may be available for recycling following decontamination, decommissioning, and dismantlement of 123 commercial power reactors is estimated to be about 650,000 metric tonnes, of which about 77% is carbon steel, 17% is stainless steel, and the remainder consists of a variety of other metals and metal alloys. This section describes the uncertainty in these estimates. **In summary, it is concluded that, based on currently available information, the estimated inventory of scrap metal potentially available for recycling from commercial nuclear power reactors is not likely to be more than a factor of two higher or lower than the estimated value.**

Quantities of scrap metals and their contamination levels used in the analysis were largely based on a deterministic model that primarily employed data from two reference reactor facilities. For this approach, uncertainty analysis of modeled estimates is best achieved by first identifying major differences among the 123 reactor plants. Differences among U.S. reactors that are deemed critical to future quantities of scrap metal involve those that define a facility in terms of its (1) physical design, (2) plant operations, and (3) choice of decommissioning alternatives. A model that adequately accounts for the variability of these parameters within the universe of reactors can be expected to yield estimates that can be viewed with reasonable confidence.

Physical Design

Foremost in defining potential scrap metal quantities are physical parameters that are determined by the class of reactor, reactor size, and period of construction. These physical variables are well documented for the 123 reactor units and were factored into scrap metal estimates by: (1) employing a Reference plant for each of the two major reactor types (i.e., Reference BWR and Reference PWR), (2) using an empirical scaling factor where plant power rating served as a surrogate measure of reactor size, and (3) using Reference facilities constructed about midway through the 30-40 year construction period that defines the nuclear power industry. (One Reference facility was pre-TMI-2 era and the other was post-TMI-2 era).

Operational Factors

Contamination levels and relative radionuclide composition on interior and exterior metal surfaces of reactors are strongly influenced by numerous operational factors. These include: (1) years of operation, (2) coolant chemistry and corrosion control, (3) fuel integrity, (4) performance or failures of critical reactor components/systems and their maintenance, and (5) health physics practices and routine cleanup efforts.

Operational factors are not, however, easily incorporated into models and therefore were not considered. This is due in part to the sporadic occurrence of some operational factors (e.g., system/component failure, fuel leakage) and in part to the subjective nature of others (e.g., quality of coolant water chemistry, corrosion control; health physics practices, etc.). Although operational factors undoubtedly contribute to large variability in the metal contamination characteristics among facilities, the individual plant differences will likely average the variabilities so that the uncertainty in the collective characteristics of the industry as a whole is believed to be relatively small compared to the individual variability among plants.

To reflect the high degree of variability as reported by studies referenced in Appendix A and a small number of current decommissioning plans, plant systems in this report were grouped into one of three levels of contamination, where each level represents a range of values that spans three or more orders of magnitude.

The projected contamination levels used in this report may represent upper-bound values. This is due to the biased data from which contamination levels were necessarily derived. The studies that were used to derive contamination levels (e.g., NUREG/CR-4289) present data from reactor facilities with abnormal histories of operation and which are not representative of the industry at large. Most of the operations at these reactors preceded the 1979 TMI-2 accident and reflect material composition, plant systems, and operational standards of the pre-TMI era. The 1979 accident triggered major reforms in the commercial nuclear industry in the form of more stringent Federal regulations and performance standards issued by the NRC. Post-TMI reforms also reflect the introduction of new standards, guidance, recommendations, and good practices from the American National Standard Institute (ANSI), American Nuclear Society (ANS), American Society for Testing and Materials (ASTM), National Council on Radiation Protection and Measurements (NCRP), Electric Power Research Institute (EPRI), and others. The organization that has most significantly influenced post-TMI plant operations is the Institute of Nuclear Power Operations (INPO). INPO's efforts to improve and standardize reactor plant operations can be expected to have two principal effects on reactor contamination levels at the time of decommissioning. On average, contamination levels can be expected to be below those identified in this report and the range or variability of contamination levels among individual plants is likely to diminish.

Decommissioning Alternatives

The final variable affecting scrap metal contamination levels (and consequently scrap metal quantities) is the choice of available decommissioning alternatives (DECON, SAFSTOR, or ENTOMB). SAFSTOR, with its extended delay in dismantling/decommissioning, will have the obvious impact of reducing contamination levels by up to several orders of magnitude. The ENTOMB option is not expected to be used.

Depending on prevailing decontamination technologies and economic factors, a reduction in residual contamination levels in scrap could significantly increase scrap metal quantities. For example, if prevailing cost-effective decontamination technologies were limited to reducing contamination to four orders of magnitude, scrap metal at 10-years post-shutdown with activity levels > 5×10^7 dpm/100 cm^2 could not be expected to meet the current release standard of 5,000 dpm/100 cm^2 and would, therefore, be excluded from recycling. Under the SAFSTOR alternative, if starting contamination levels were reduced by several orders of magnitude through natural decay, an expanded fraction of the total pool of scrap metal could be expected to meet potential release criteria.

At this time, however, the vast majority of reactor licensees have not revealed their preference for a specific decommissioning alternative and speculation regarding decontamination technologies for nearly a century into the future would be unwise. For these reasons, uncertainties associated with decommissioning alternatives were not addressed in this report. Scrap metal quantities and residual contamination levels were based on a 10-year post-reactor shutdown period and current decontamination technologies.

Summary Conclusions Regarding Uncertainty

There is significant diversity among the current U.S. inventory of 123 licensed nuclear reactors that is likely to yield variable quantities and contamination levels of scrap metal among individual reactor units at the time of decommissioning. Modeled estimates based largely on two reference facilities support the following statements:

(1) Physical differences inclusive of plant design, power rating, and period of construction are thought to be the most important parameters affecting scrap metal quantities for individual reactors and were incorporated into the modeled results.

(2) Parameters that could not be readily defined (i.e., operational factors) are likely to represent a continuum with a symmetrical distribution about a mean value. Thus, factors contributing to low quantities of scrap metal containing radioactive contamination for some plants will be offset by others yielding higher than expected scrap metal quantities. As such, the uncertainty in the collective quantities and radionuclide inventories for all plants combined are likely to be considerably smaller than the variability among plants.

(3) Variations among reactor plants concerning selection of a decommissioning alternative will likely impact residual contamination levels of individual reactor systems. Although residual contamination levels undoubtedly affect decontamination strategies and costs, the mass quantity of available scrap may only be modestly affected.

(4) Based on currently available information, it is concluded that the estimated collective inventory of scrap metal available for recycling from commercial nuclear power reactors is not likely to vary by more than a factor of two higher or two lower than the estimated value of 650,000 metric tonnes.

10.2.2 Scrap Metal from DOE Facilities

In this report, scrap metal estimates for DOE facilities were defined as: (1) existing scrap metal inventories that are currently stored at 13 DOE sites and (2) future scrap metal inventories that are anticipated as a result of decommissioning. The estimated existing inventory of about 171,000 metric tonnes of existing scrap is small in comparison to future quantities which are projected to exceed 925,000 metric tonnes. **Based on available data, the current estimated value of existing DOE contaminated scrap is considered not likely to differ from the true**

value by more than a factor of two. The quantity of scrap associated with the future decommissioning of DOE facilities is more uncertain and may be several times higher than the estimated value. The actual future quantity could also be perhaps a factor of two lower if many of the DOE facilities are decontaminated and used for other purposes.

As will become apparent in the discussion that follows, these estimates of uncertainty reflect the judgment of the authors based on a review of the available literature and are provided as a basis for further discussion and future investigations.

Table 10-1 identifies site-specific scrap metal quantities and primary source documents from which data were obtained to estimate scrap quantities. The information is limited to deterministic (i.e., single) values of scrap metal estimates at individual DOE sites, with no additional data that would further define accuracy or variability of cited values. Moreover, available data were frequently speculative, incomplete, or insufficiently detailed. Quantitative and qualitative deficiencies in available data, therefore, necessitated the use of surrogate values, assumptions, and interpolation.

Table 10-1. Selection of Data Sources for Scrap Metal Quantities at DOE Facilities

DOE Site	Existing scrap metal (MTs)		Future scrap metal (MTs)		
	Source Document		Source Document		
	MIN 96	HAZ 95	MIN 96	EPA/SCA 95	DOE 95
Fernald	4,218			135,623	
Hanford	377			91,798	
INEL	727		33,486		
LANL		3,099		2,686	
NTS	264		—	—	—
ORNL	1,129		—	—	—
Y-12	9,065		—	—	—
K-25	29,357				212,706
Paducah	48,374				230,886
Portsmouth	8,914				189,072
Rocky Flats		24,543		26,303	
SRS	13,183		3,054		
Weldon Spring		27,839	—	—	—
SubTotal	171,089		925,614		
TOTAL	1,096,703				

In acknowledgement of data limitations and their adverse impact on uncertainty, the DOE stated the following:

> ". . . Because of limited data, this report does not attempt to capture the exact amount of each material in inventory. Rather, it attempts to capture the general magnitude of the inventory of each material (MIN 96)."

Elsewhere, the DOE (MIN 96) concluded that while the ". . . Department maintains detailed inventory systems of weapons components . . . there is no reliable system to identify a complete inventory of scrap metal and equipment (MIN 96)." (Emphasis added)

A reasonable interpretation of these statements is that cited quantities reflect best estimates (as opposed to comprehensive measurements) and, therefore, pose a significant but undefined level of uncertainty. As a rule, deterministic data preclude a rigorous uncertainty analysis. Reliability of deterministic data, however, may be assessed by means of a subjective evaluation that focuses on the methods employed by DOE for data collection.

DOE Data-Collection Methods

DOE's 1996 Materials in Inventory Initiative was a year-long Department-wide effort aimed to improve management and disposition for materials that may no longer be needed. The objectives of this effort focused on management approaches for: (1) uncontaminated materials, (2) suspect materials, (3) contaminated material, and (4) clearing suspect or contaminated materials to the property management system for reuse or release. Data sources for the MIN Scrap Metal and Equipment Team that developed scrap metal estimates included the following:

- DOE Regulations, Policies, and Orders that served as sources of information on requirements and procedures for managing scrap metal and equipment.
- Studies conducted within the past five years pertaining to scrap metal inventories within the DOE complex.
- Information collected in response to surveys, site-visits, and national conferences.

Useable data were defined for 13 sites with significant scrap metal inventories. Data-collection methods for individual sites varied, however; while some sites have developed and maintained databases that are current for scrap-metal inventories, others relied exclusively on historical knowledge to determine quantities of scrap under their control.

With regard to the reliability of collected data, the MIN Scrap Metal and Equipment Team offered the following statement:

> " . . . Data limitations include the following: (1) no information was received on either scrap or equipment for [several DOE facilities] . . . (2) some information was submitted in summary form only, without site-specific breakouts; (3) some sites supplied complete information on some topics and partial or no information on others ... and (4) some data could not be tabulated because it was descriptive rather than quantitative or expressed in units inconsistent with the units used in this report and could not be readily converted (MIN 96)."

Noteworthy is the DOE's reference to "partial or no information" that shows that only about one-fifth of the total scrap-metal inventory had been assessed for the presence of radioactive contamination. In other words, four-fifths (80%) of existing metal inventories have not been assessed for radioactive contamination. About 88% of the assessed fraction was determined to be contaminated with radioactivity. This relationship was used to estimate the percentage of contaminated scrap within the unassessed fraction of existing metal scrap.

In summary, the collective uncertainty of existing scrap metal quantities reflects the combination of uncertainties contributed by the following:

(1) the uncertainty in total scrap metal estimates as reported by individual DOE sites that in some instances were based solely on historical records;

(2) the large percentage of scrap (about 80%) that was "unspecified" with regard to radioactive contamination and the resultant need to apply a scaling factor derived from the 20% of scrap that had been assessed for contamination; and

(3) the variability of existing scrap metal inventories as a function of time.

With regard to the third component of uncertainty, most sites reporting data for the MIN initiative indicated that their inventories of existing scrap may be sold or otherwise dispositioned on a routine basis. The extent of variation in inventories with time can, therefore, not be assessed from the snapshot of inventories as currently reported.

On the basis of available data, the current estimated value of 171,000 metric tonnes of contaminated scrap is considered not likely to differ from the true value by more than a factor of two. Thus, lower- and upper-bound values of existing contaminated scrap metal are defined to be 85,500 and 342,000 metric tonnes, respectively.

Uncertainties Regarding Future Quantities of Scrap Metal

Of the 13 sites with existing scrap-metal inventories, current data identified only nine sites as future sources of scrap metal estimated to be about 925,000 metric tonnes (Table 10-1).

For individual DOE sites, point estimates were largely derived from historical data pertaining to design specifications of buildings, structures, and process equipment that have been slated for decommissioning.

The level of uncertainty regarding future quantities of scrap metal is undoubtedly higher than that of existing scrap-metal quantities. Compounding the shared uncertainty of simply quantifying a known or suspected aggregate of metal components is the incomplete and uncertain decommissioning schedule on which future scrap metal estimates are based.

Assumptions regarding future political, social, and economic factors that may significantly impact the current decommissioning schedule cannot readily be factored into a discussion of uncertainty. A reduced scope of decommissioning activity is likely to result in future scrap metal quantities less than the currently-estimated value of 925,000 metric tonnes. Conversely, an expanded decommissioning that extends beyond the nine DOE sites defined in this report would be expected to significantly increase the projected scrap quantities beyond current estimates.

10.3 UNCERTAINTY FOR NORMALIZED RMEI DOSES AND RISKS

Normalized doses and risks for the RMEI are provided in Chapter 7 for a total of 40 radionuclides. Normalized dose values are reported in units of mrem/yr per pCi/g activity concentration in released scrap metal and normalized risks are defined in units of lifetime cancer risk per pCi/g activity concentration in released scrap metal. **In general, the analyses demonstrate that the normalized doses for the RMEI could be higher by a factor of 5 to 50, or lower by up to a factor of 100 to 500, depending on the radionuclide. The uncertainties in the normalized risks are similar, except that the possibility exists that the risks could be zero for extremely low doses and dose rates.**

When scrap metal containing nominal levels of residual radioactivity is released for unrestricted recycling, human exposure may occur at discrete stages of the life cycle of scrap. At each stage of the life cycle, exposure may be dominated by select radionuclides and exposure pathways that affect certain individuals within the exposed population more than others. In preceding chapters, dose estimates were derived by evaluating potential exposures for all life-cycle stages of scrap and associated groups of exposed individuals. Population groups that were found to have the highest normalized dose for a given radionuclide provided the basis for modeling the RMEI. Selected as RMEIs are individuals within each category of exposed individuals who, on the basis of reasonable assumptions, could be expected to receive doses toward the high end of the distribution of doses for members of each category. Reasonable assumptions required selecting values or a range of values for specific model parameters.

The uncertainty in the normalized dose to the RMEI is principally due to the wide range of potential values that may conceivably characterize each of several critical model parameters used. Table 10-2 provides a summary of the uncertainties for select groupings of radionuclides that are considered most limiting to RMEI exposures. For each radionuclide grouping, the critical stage within the life cycle of scrap is identified along with dominant exposure pathway(s), and an estimate of the "upper-end multiplier" and "lower-end divisor" is provided. The upper-end multiplier and lower-end divisor define the potential range (and therefore the uncertainty) of the normalized RMEI doses. For example, Table 7-1 in Chapter 7 previously identified the RMEI dose of 0.899 mrem/yr per pCi/g of Co-60 in released scrap metal. This dose was estimated for a worker operating a metal lathe fabricated totally from scrap metal containing the Co-60 contaminant at a concentration of 1 pCi/g. According to Table 10-2, this normalized dose could be as much as a factor of five greater if the "metal product" was assumed much larger than the modeled lathe and more time was spent by the individual in close proximity to the product. Conversely, the normalized dose could be lower by as much as a factor of 100 if (1) a smaller metal product, (2) shorter exposure times, and (3) a variable percentage (i.e., less than 100%) of contaminated scrap used to produce the metal product were assumed.

The multipliers and divisors are largely based on professional judgment and are designed to bracket estimated uncertainties and variabilities for normalized RMEI doses. Detailed explanations for upper-end multipliers and lower-end divisors for each of the six limiting life-cycle stages identified in Table 10-2 are provided in Section 5.4.7 of Appendix L.

A summary of multipliers and divisors that were defined for all 40 radionuclides is displayed in Figure 10-2 in the form of a bar chart. For most radionuclides, the range of uncertainty, as defined by lower- and upper-bound values, spans about three to four orders of magnitude. Normalized values for the RMEI are quantitatively defined by the upper end of the distribution of values divided by the line that marks the boundary between upper- and lower-bound values.

Table 10-2. Uncertainty/Variability in Normalized Individual Doses

Radionuclides	Limiting Stage	Primary Pathway	Upper End Multiplier	Lower End Divisor	Bases
Zn-65* Sb-125 Cs-134* Cs-137*	Scrap yard	External exposure	10	100	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30 fold), reduced occupancy and increased distance (3).
Ni-59 Ni-63 Mo-93 Tc-99 Ac-227+D		Inhalation	10	500	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30 fold), reduced occupancy (2), and reduced dust loading (10)
Fe-55		Soot ingestion	10	500	Upper end due to eliminating dilution factor. Lower end due to additional dilution (30), reduced occupancy (2), and reduced soot ingestion (10)
Mn-54 Co-60 Ru-106 Ag-110m+D	Metal products	External exposure	5	100	Upper end due to increase in size of component and occupancy time (5). Lower end due to application of a dilution factor (30) and lower occupancy time and smaller size component (3).
Nb-94 Ce-144+D Eu-152 Ra-226+D Ra-228+D Th-228+D	Slag pile	External exposure	40	100	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (4). Lower end due to additional dilution (30) and smaller contaminated area and occupancy time (3).
Pm-147 Th-229/230/232 Pa-231 U-234/235/238 Np-237 Pu-all Am-241 Cm-244	Slag pile	Inhalation	20	500	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (2). Lower end due to additional dilution (30), lower dust loading (10), and lower occupancy time (2).
Pb-210	Mill	Ingestion	20	500	Upper end due to elimination of dilution factor (8) and increased occupancy time and slag partition (2). Lower end due to additional dilution (30), lower soot ingestion (10), and lower occupancy time (2).
C-14	Offsite exposure to airborne emissions	Ingestion of food	50	NA†	Upper end due to elimination of dilution factor (8), closer location (3), increased intake of crops (2).
I-129				100	Lower end due to additional dilution (30), further distance (2), less intake (2).
Sr-90	Ground water contaminated by slag leachate	Ground water ingestion	50	NA†	Upper end due to less dilution in ground water. Lower end due to elimination of ground water due to increased transit time, and soot ingestion becomes the limiting pathway.

* These radionuclides partition to baghouse dust. If it is plausible for individuals to be exposed to reconcentrated stages of the metal recovery process for prolonged periods of time, the upper end multiplier for these radionuclides could be as high as a factor of 100.

† A lower limit for these pathways in not applicable, since the lowest limiting dose will be due to a different pathway (see text).

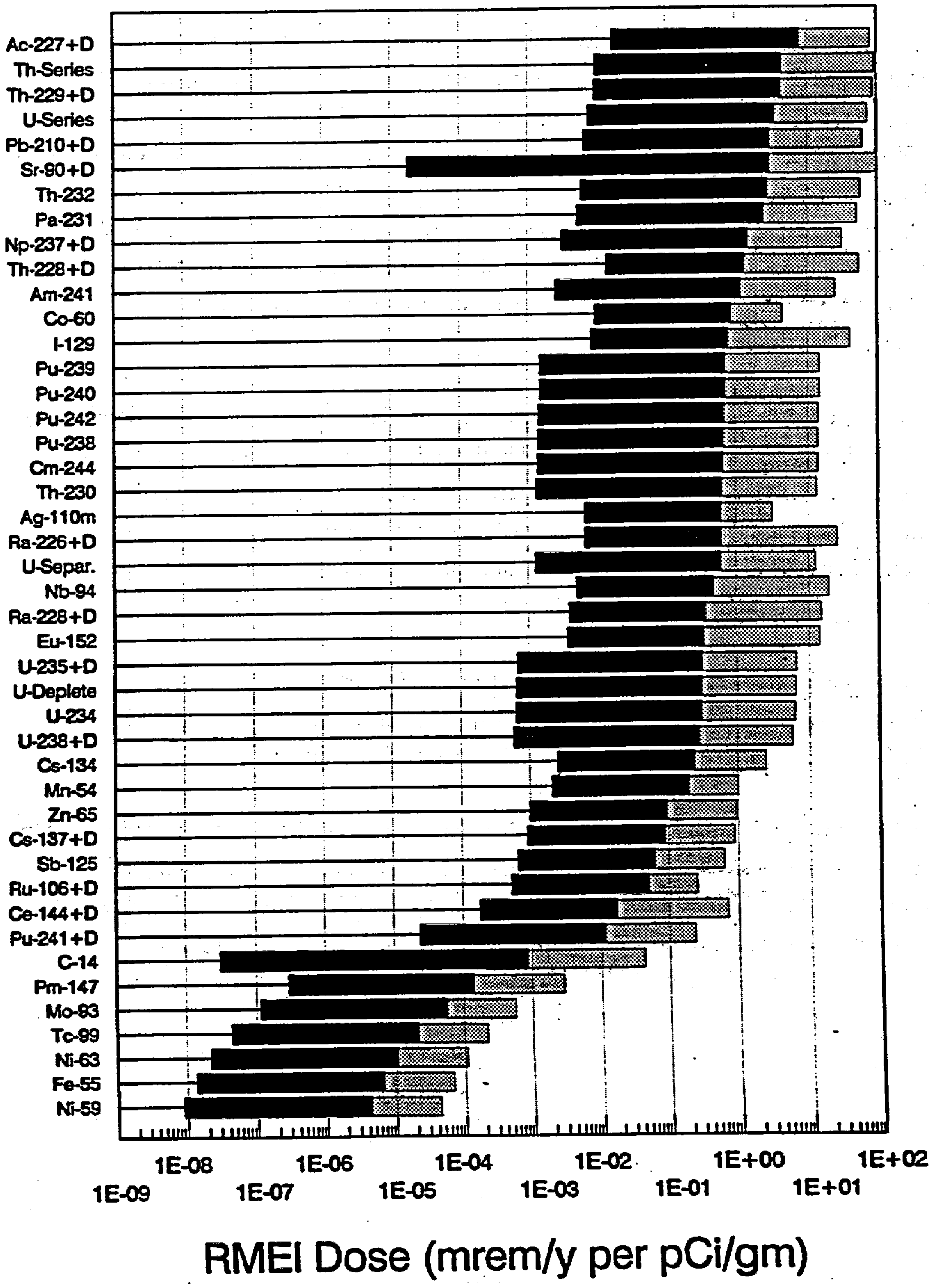

Figure 10-2. Bounding Normalized RMEI Dose Values (mrem/y per pCi/g)

10.4 UNCERTAINTY IN NORMALIZED COLLECTIVE DOSE ESTIMATES

Normalized collective doses were derived in the preceding chapter and are reported as radionuclide-specific values in units of person-rem per Ci in released scrap. Important to note is that the normalized collective dose does not contain a measure of time. The normalized population dose represents the sum of all individual exposures for the entire exposed population, for as long as the radionuclide can reasonably be assumed to result in human exposures. **In general, the results of the uncertainty analyses reveal that the collective doses could be higher or lower than the estimates by a factor of between 2 and 3. The collective risks could be higher by the same amount, but the possibility exists that the risks could be zero for extremely low doses and dose rates.**

When contaminated scrap metal is recycled, some radionuclide contaminants are predominantly (and in some instances exclusively) partitioned in the finished metal from which new products are made; based on chemical and physical properties, other radionuclides are more likely to be found in slag, baghouse dust, or entrained in stack gases that are emitted into the ambient biosphere. Figure 10-3 identifies each of the four dominant media associated with population exposures and their most limiting radionuclides.

Radionuclides contained in the four media have the potential for exposing large numbers of individuals to doses that will vary from near zero up to the dose defined for the RMEI. Individual doses may, therefore, span four or more orders of magnitude.

A second important characteristic of the collective dose, and one which has great significance to the discussion of uncertainty, is the fact that large (and undefined) variations among individual exposures have little or no impact on the magnitude and variability of the collective dose. In effect, most model parameters that define individual doses and their variability are inconsequential to modeling the collective dose of an exposed population.

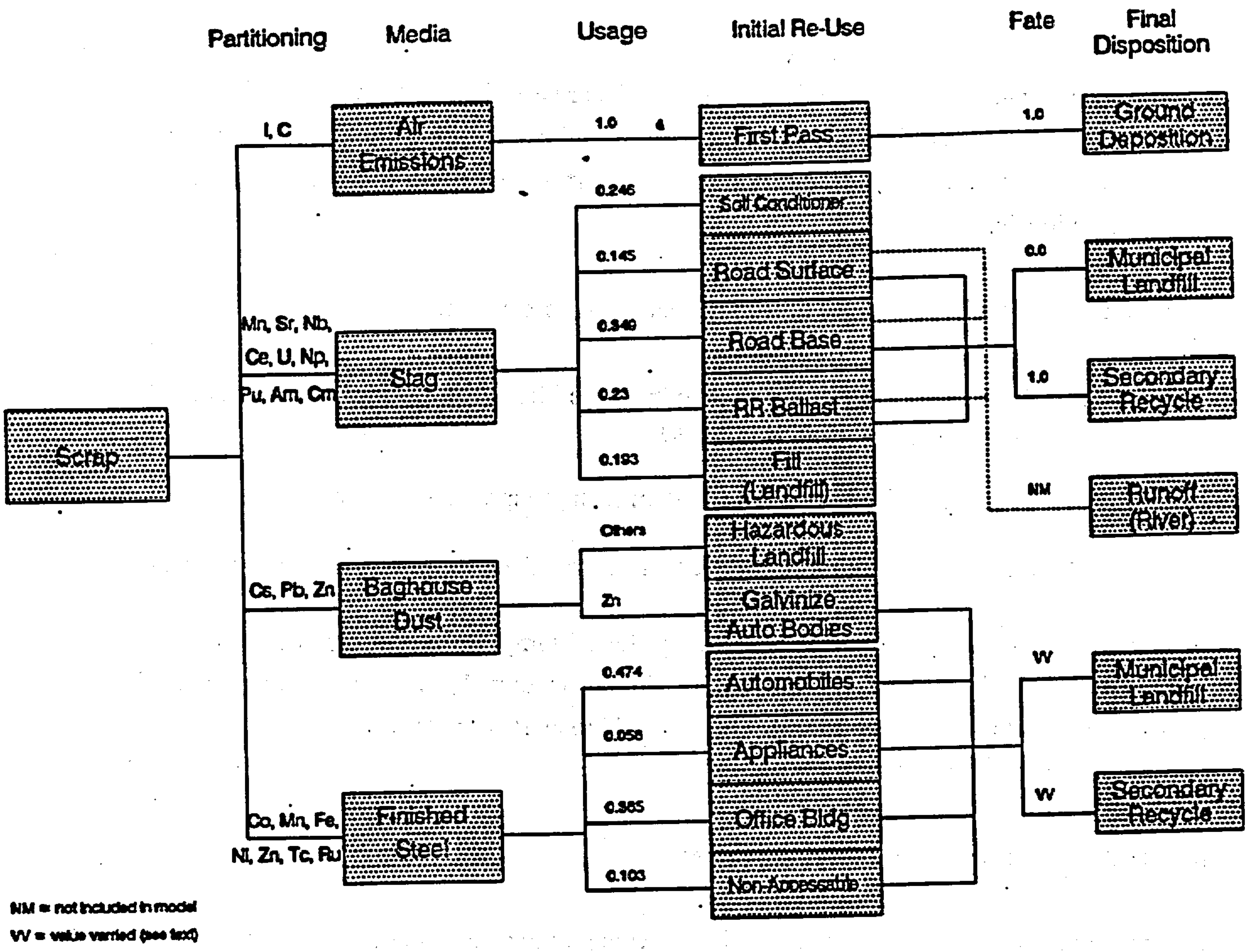

Figure 10-3. Collective Impact Calculational Approach

This important generic feature of normalized collective dose is readily illustrated by the following example:

> From Figure 10-3, carbon-14 is identified to partition to flue gases that may be released to air.
>
> For Case #1, it is assumed that a one-Curie quantity of C-14 is released from a smelter stack located in a residential community under very stable atmospheric conditions with minimum dilution. In turn, this results in maximum air concentrations to the downwind population. However, offsetting the high air concentrations and maximally elevated individual doses is the fact that the total number of exposed individuals is at a minimum.

For Case #2, the same one-Curie quantity when released under highly unstable atmospheric conditions results in maximum dilution. Correspondingly, this results in greatly reduced individual doses, but to a proportionately larger number of exposed individuals.

For the two contrasting cases, the following conclusions are drawn:

(1) the average individual exposure in each of the two groups will be different (perhaps by orders of magnitude);

(2) the range of individual doses within each group will differ substantially; but

(3) the normalized collective dose will be essentially the same for both conditions.

Since only a restricted number of parameters significantly impact the estimated normalized collective dose, its level of uncertainty is proportionally restricted. Thus, the several orders of magnitude spread in the RMEI dose estimate is likely to be reduced to just one order of magnitude for collective dose estimates.

Model Parameters that Impact Uncertainty

Critical parameters that are likely to introduce uncertainty to collective dose estimates were previously identified in Figure 10-3 and include the following:

(1) Partitioning Factors. For each radionuclide contained in scrap, fractional values were assigned to each of the four media based on physical and chemical properties.

(2) Usage Factors. With the exception of "air emissions," available data were assessed to best determine the various uses for each media. For example, it was determined that about 34.9% of slag is currently used for road base, 24.6% for soil conditioner, 23% for railroad ballast, etc.

(3) Initial Re-Use. This identifies specific exposure scenarios for which additional input parameters must be defined, as will be explained below.

(4) Fate and Final Disposition. Initial re-use in some instances may not fully define the collective dose if the radionuclide persists beyond the useful life of the object and the object is recycled again. Thus, an automobile that is retired after 10 years may again be recycled. Due to natural decay, residual contamination may be greatly reduced, but may nevertheless remain significant.

When combined, the variability of parameters needed to model normalized collective dose is not likely to yield uncertainties greater than a factor of 10. To illustrate the types of parameters used for modeling and their variabilities, the following illustration is offered.

Model Parameters Used to Estimate Normalized Collective Dose for Co-60

Partition Coefficient. Cobalt has chemical and physical properties that closely resemble iron. A review of the scientific literature confirms the fact that, as a contaminant in scrap metal, Co-60 is retained nearly 100% in the metal melt. For Co-60, the assigned partition coefficient of 100% to finished steel is, therefore, considered to have an uncertainty approaching zero.

Usage. Steel is used to make a wide variety of finished products; an exhaustive analysis of usage is not practical. Data provided by the steel industry were used to define principal categories and markets that employ recycled steel. From current data, information was gleaned that indicates: (1) nearly half (i.e., 47.4%) of recycled scrap is used to produce automobiles; (2) 36.5% goes to the production of structural components; (3) 5.8% is consumed in the manufacturing of consumer items inclusive of home appliances; and (4) 10.3% is contained in items that are "nonaccessible" and will not contribute to human exposure. The uncertainty in these values is defined by the potential that current usage or appropriation of scrap metal may change in the future for reasons such as a shift in national/global economics; new technologies pertaining to steel production and manufacturing process; and substitution of steel for other metals/alloys or non-metallic materials.

Since there are major economic commitments to the steel industry and associated manufacturing processes for consumer items, a significant deviation from current uses of scrap metal is considered unlikely. The uncertainty in model usage factors is, therefore, considered insignificant.

Initial Re-Use Exposure Scenarios

To develop normalized collective dose estimates, scenarios were required that characterize individuals and their typical use of or exposure to automobile(s), home appliances, and office building(s) (see Figure 10-3). These scenarios were described in detail in Chapter 9, Section 9.5, and are briefly summarized here.

Automobile Scenario. The weighted average physical mass, dimensions, and density of primary steel components that include engine, frame, and shell were derived from current data of automobiles manufactured in the United States. These data provided the input parameters for the computer code MicroShield™ for the calculation of exposure rate to an occupant. (MicroShield™ is considered a standard in the industry and can be assumed to yield reliable results for a given set of input parameters.) Average automobile occupancy was estimated at 1,460 person-hours per year and was based on two occupants for two hours every day. The automobile's effective life was assumed to be 7.3 years; however, it was further assumed that it would be recycled and 100% of its scrap would be returned to the pool of scrap used to manufacture automobiles for a period of 1,000 years.

For the free release of scrap containing one-Curie of Co-60, the automobile scenario yielded a normalized collective dose of 6,606 person-rem. Uncertainty of this modeled output value is principally limited to the accuracy of computer-generated (i.e., MicroShield™) dose rates and the assumption regarding automobile occupancy. Of the two, automobile occupancy is likely to be the larger contributor to the collective uncertainty. The analysis assumes an occupancy of two persons per vehicle driving 730 hours per year. The analysis further assumes 730 hours of driving per year at an average speed of 35 mph which results in an average yearly mileage of about 25,500 miles per vehicle. This is more than twice the national yearly average of 11,834 miles as reported for 1993 by the U.S. Department of Transportation and the Insurance Institute for Highway Safety. Accordingly, the estimated collective dose of 6,606 person-rem resulting from automobile use is likely to be conservative with an uncertainty of about a factor of two.

Appliance Scenario. Key home appliances with the potential for human exposure include refrigerators, stoves, dishwashers, microwave ovens, trash compactors, washers, and dryers. These items are commonly located in or near the kitchen, where members of a household spend considerable amounts of time. The average exposure dose rate for this scenario was modeled by means of the MicroShield™ computer code with input parameters that included average mass of appliances, shielding factors, and average physical dimensions that define a kitchen/dining area.

To estimate kitchen occupancy times, a family of four was assumed. Furthermore, it was assumed that all members of the family eat breakfast in the kitchen seven days a week, and eat dinner in the kitchen five days a week. Weekday lunch was assumed to be eaten in the kitchen by only one member of the family. It was also assumed that one member of the family would spend one hour each week night doing homework in the kitchen. Based on these assumptions, kitchen occupancy times of 70 person-minutes per day in the work area and 190 person-minutes per day in the dining area were calculated.

The normalized collective dose was estimated to be 340 person-rem. (This also includes 30 person-rem from cookware.) The uncertainty of this value is governed primarily by the 70 and 190 person-minutes occupancy times that were assumed typical for U.S. households. As average values, these occupancy estimates are not likely to be more than a factor of two (multiplied or divided by two) different from actual current (and future) values.

<u>Office Building Scenario</u>. An office building was selected to best represent the exposure scenario from finished steel used in structural building components. The model office building was of modular design with six offices in each module. Dose rates from structural components consisting of I-beams, steel columns, studs, rebar, etc. were again estimated by means of the MicroShield™ computer code. Based on an occupancy factor of 2,000 hours per year (per individual), 2.8 persons per 100 m^2 of building floor space, and a 50-year building service life, a normalized collective dose of 3,250 person-rem was calculated.

The uncertainty in this estimate is considered to be small but the dose estimate can be considered to be conservative because the critical model parameters included the standard 2,000 hours per year exposure period and the 2.8 persons per 100 m^2 worker density values. The latter value was obtained from the Commercial Building Energy Consumption Survey (CBECS) published by the Energy Information Agency of the Department of Energy. The survey, however, also reported human population densities for the following:

- all commercial buildings: 1.04 persons per 100 m^2,
- buildings used for lodging (hotels/motels): 0.82 person per 100 m^2.

A building exposure scenario that represents a weighted average of all facilities might have reduced the normalized collective dose by about a factor of two.

Summary Conclusions

Uncertainties in normalized collective doses derived in the TSD are believed to be relatively small. The basis for this generic assumption is two-fold:

(1) The number and complexity of parameters needed to model collective dose are few in comparison to those required to model individual doses (e.g., RMEI).

(2) Collective model parameters are defined by values that are robust since they reflect average values that in most instances are readily defined in the literature.

In the above-cited example, the free release of scrap metal containing one-Curie of Co-60 was estimated to yield a normalized collective dose of 1.02E+04 person-rem. This estimate was based on the following contributions:

Source/Scenario		Dose (person-rem)	Estimated Uncertainty Factor
Automobile		6.61E+03	2 lower and less than 2 higher
Appliances (and cookware)		3.40E+02	< 2 higher and lower
Office Building		3.25E+03	2 - 3 higher and lower
	Total	1.02E+04	about 2 higher and 3 lower

Analysis of principal model parameters for each of the three scenarios suggests a modest level of conservatism that is unlikely to yield a collective dose uncertainty greater than a factor of two higher or three lower.

10.5 UNCERTAINTIES REGARDING MINIMAL DETECTABLE CONCENTRATIONS FOR RADIONUCLIDE CONTAMINANTS

Demonstration of compliance with any future radiation protection standards governing the release of scrap metal from nuclear facilities will require radiological measurements, most of which involve use of field survey instruments.

Whether a particular instrument and measurement procedure is capable of detecting residual activity at a certain fraction of a future regulatory limits is largely determined by its minimum detectable concentration (MDC) values.

Chapter 8 of the TSD discusses the standard instrumentation that may be used in surveys for the free release of scrap metal and identifies the following generic equations that are used to calculate MDC values:

- <u>Direct Measurement (Fixed Position)</u>

  The MDC for direct measurements is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{t}{60}}}{\frac{t}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100}} \qquad \text{(Eq. 10-1)}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
t = count time (s)
60 = conversion factor (s/min)
Y_i = yield for emission I (ptcle-emitted/d)
ϵ_i = detector efficiency for emission I (c/ptcle-emitted)
A = detector area (cm^2).

- <u>Surface Scanning</u>

  The MDC for detection of large areas of contamination using surface scanning is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{2 * \tau}{60}}}{\frac{2 * \tau}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100} * HF}$$

(Eq. 10-2)

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
t = meter time constant (s)
60 = conversion factor (s/min)
Y_i = yield for emission I (ptcle-emitted/d)
ϵ_i = detector efficiency for emission I (c/ptcle-emitted)
A = detector area (cm^2)
HF = surveyor efficiency (%).

Inspection of Equations 10-1 and 10-2 identifies parameters that can be termed as either "intrinsic" or "extrinsic." Intrinsic parameters affecting MDCs are defined by the manufacturer's design of the instrument and include detector dimensions, window thickness, meter time constant, and other design aspects that affect detection efficiency. Based on reasonable quality assurance standards by the manufacturer and compliance with stated instrument specifications, these parameters of MDC are unlikely to significantly impact the uncertainty of the calculated MDC value.

In general, calculated MDCs, inclusive of those cited in the literature or specified by the manufacturer, represent optimum instrument capabilities and instrument use under controlled laboratory conditions. The uncertainty of a calculated MDC value must, therefore, be defined by extrinsic differences that define controlled laboratory measurements and field survey measurements.

Extrinsic factors can be further categorized as operational parameters and conditional parameters. Critical operational parameters include the following:

- Instrument Selection and Radionuclide Identification. Survey instruments must be selected that properly reflect the radionuclide contaminants that are most likely to be encountered.

  In most cases, the identification of critical radionuclides should be straightforward since the nature of the operation at the plant is known. Half-lives of isotopes can be used to determine what may have decayed away and what is likely still to be present.

  For situations in which a fixed ratio between two radionuclides can be established throughout a cleanup unit, the measurement of one radionuclide can serve as a surrogate for the other. This might also be possible for more than two radionuclides if consistent ratios between them can be demonstrated.

- Instrument Calibration. Accurate and reliable measurements of surface contamination requires proper selection of calibration standards. Calibration sources must be selected that accurately reflect the type/mix of radiation emissions and their energies.

- Source to Detector Distance. A critical parameter affecting instrument efficiency and, therefore, MDC values is the accuracy and consistency of source to detector distance. This is especially critical for instruments used in a "scanning" mode.

- Operator Experience. Where contamination levels need to be checked over a fine spatial scale, all of the surface area should be measured by scanning, i.e., by passing a survey meter probe over the surface at a fixed rate and covering the entire area. The ability to measure a given level of radioactive contamination is, of course, affected by the detector's sensitivity, the particular radionuclide, and the subjective ability of the operator to discern a change in the reading either by visual or audible means.

- Choice of Operating Parameters. For stationary survey measurements, the duration of time for which counts are integrated is critical to the MDC. For scanning mode operations, the selection of instrument time constant and the scanning velocity are critical.

Conditional factors are those that define the survey environment. Most notable among these is the ambient radiation background level at the survey location. Equations 10-1 and 10-2 establish the relationship between detector background count rate and MDC: MDC increases in direct proportion to the square root of the ambient background count rate.

For nominal background levels ranging from 50 cpm to 100 cpm, an assumed MDC for a pancake probe is likely to be in close agreement with field-detection limits. With the realization that future scrap may have to be surveyed in radiologically controlled areas (RCAs) of a contaminated facility, ambient background levels may be well above those of a laboratory setting or assumed for the calculated MDC values. When actual ambient background count rates significantly exceed assumed background values used to derive MDCs, the potential exists for the free release of scrap metal that in fact may not meet a stated release standard. Figure 10-4 demonstrates the relationship between MDC values and ambient background.

A second conditional factor affecting MDC uncertainty involves surface materials and their texture. An *a priori* calculation of the MDC assumes uniform, smooth, and flat surfaces. For surfaces that may be pitted, corroded, and three-dimensional, conversion of the surface emission rate to estimates of residual contamination may yield values that significantly underestimate actual residual contamination.

The uncertainty introduced by operational and conditional factors cited above on MDCs that are calculated on an *a priori* basis are summarized in Table 10-3. The values represent multiplicative factors to be applied to calculated MDCs. From these data, the following conclusions should be drawn:

(1) Before any survey measurements are performed that may result in release of scrap, the survey instrument and measurement procedure to be used must be shown to possess sufficient detection capabilities relative to specified surface-contamination release limits.

(2) Based on uncertainty values cited in Table 10-3, the detection limits of the survey instrument must be a fraction of the limit that is defined by the reciprocal of the highest uncertainty value (e.g., for direct measurement of beta-emitting radionuclide, the calculated MDC value of the instrument should be less than one-third (1/3) of the regulatory release limit).

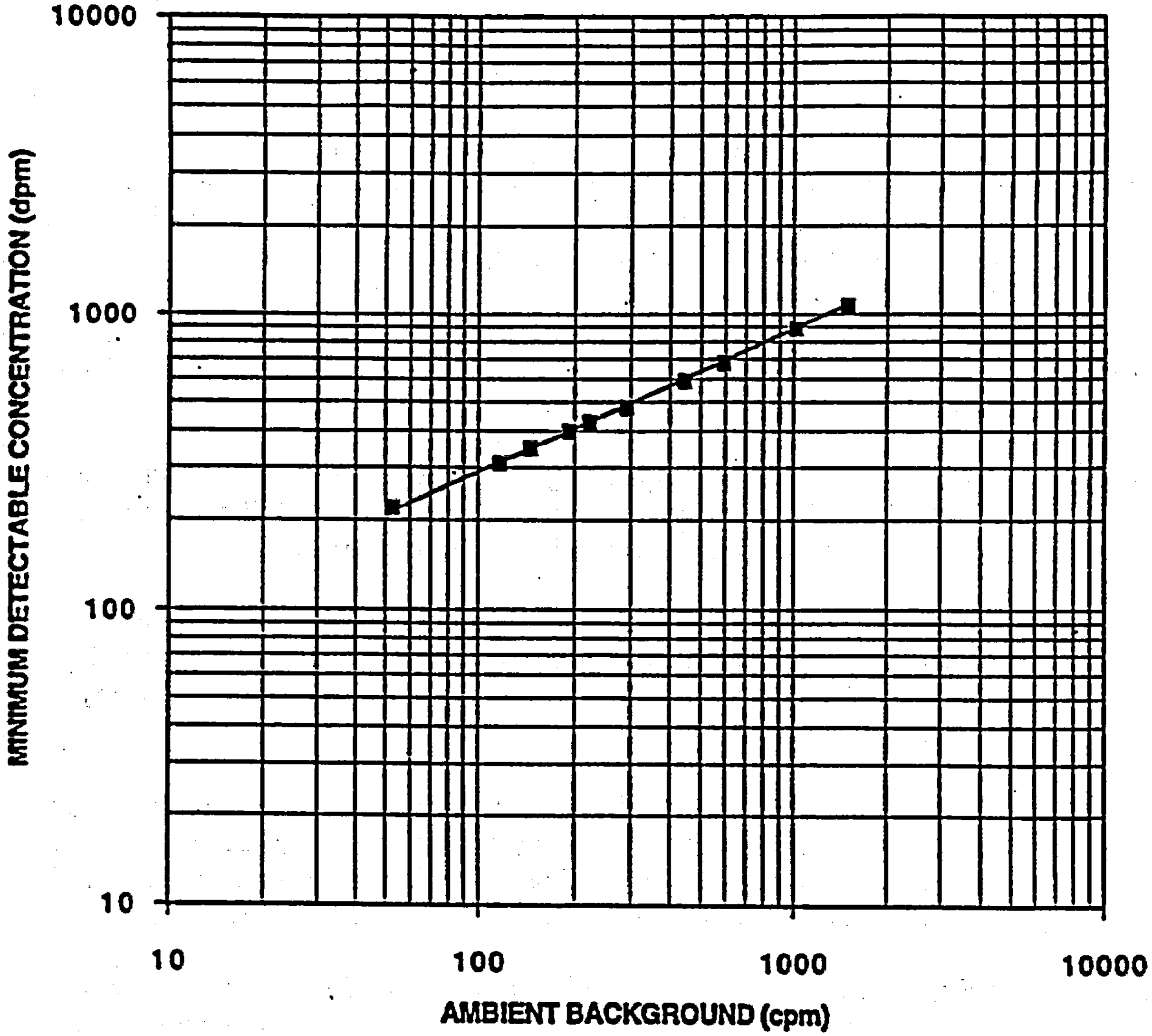

Figure 10-4. Effects of Ambient Background on MDC Calculation

Table 10-3. Relative Range in MDCs*

Survey Mode	Beta	Alpha	Gamma
Direct measurement	1 - 3	1 - 5	1 - 7
Scan - small area source	1 - 7	1 - 14	0.01 - 1
Scan - large area source	1 - 4	1 - 7	1 - 8

* The values are multipliers to be applied to the MDCs tabulated in Chapter 8.

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REVIEW DRAFT

TECHNICAL SUPPORT DOCUMENT

EVALUATION OF THE POTENTIAL FOR RECYCLING OF SCRAP METALS FROM NUCLEAR FACILITIES

VOLUME 2 OF 3: APPENDICES A-F

Prepared by:

S. Cohen & Associates, Inc.
1355 Beverly Road
McLean, Virginia 22101

Under

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Prepared for:

U.S. Environmental Protection Agency
Office of Radiation and Indoor Air
401 M Street, S.W.
Washington, D.C. 20460

Martin Offutt
Work Assignment Manager

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VOLUME 2

APPENDICES A-F

Contents

5.0 METAL INVENTORIES SUITABLE FOR RECYCLING

From data presented in previous sections, two important conclusions can be stated: (1) only a fraction of metal inventories is likely to be significantly contaminated and (2) not all contaminated metal inventories are potentially suitable for recycling. "Suitability for recycling" is largely determined by the practicality and efficacy with which contaminated scrap can be decontaminated to a level considered acceptable in terms of risks to human health and the environment.

The choice of available decontamination methods needed to render scrap metal suitable for recycling (or unrestricted use) is largely dependent on the starting level of contamination encountered, the type of surface, physical accessibility to the surface, radionuclides involved and their chemical state(s), and size/configuration of the object requiring decontamination.

Several techniques are currently used in decontamination efforts at nuclear facilities. Their applicability, however, is not without restrictions and for nearly all approaches, there are numerous factors that affect their efficiency. Examples include the choice of cleaner/solvent/surfactant for "hand wiping"; the selection of chemical solvent for the dissolution and removal of radioactive corrosion films or base metal; or the innovative use of dry-ice (CO_2) pellets for abrasive blasting. These techniques and their general applicability and limitations are briefly summarized below.

Hand Wiping. Rags moistened with water or a solvent such as acetone can be an effective decontamination process. Wiping can be used extensively and effectively on smaller items with low-to-medium external contamination levels and easily accessible internal contamination. This method may not work well if the item is rusty or pitted. It requires access to all surfaces to be cleaned, is a relatively slow procedure, and its hands-on nature can lead to high personnel exposure. On the positive side, wiping can provide a high decontamination factor (DF), generates easily handled decontamination wastes (contaminated rags), requires no special equipment, and can be used selectively on portions of the component.

Steam Cleaning. This may be performed either remotely in a spray booth or directly by decontamination personnel using some type of hand-held wand arrangement. In the former case, only minimal internal decontamination is possible; however, reasonable external cleaning can be accomplished quickly with low exposure expenditures. Containment of the generated wastes and protection of personnel from radioactive contamination may be difficult.

Abrasive Blasting. This is a highly effective procedure even for surfaces that are rusty or pitted. As with hand-held steam cleaning, this method suffers from internal accessibility problems. It also generates large amounts of solid wastes and, being a dry process, produces significant quantities of airborne radioactivity. Abrasive blasting may be used if its high effectiveness can be justified after taking the exposure, waste, and accessibility limitations into account. Some of the aforementioned disadvantages are eliminated when dry ice (CO_2) pellets are used.

Hydrolasing. The use of high pressure water jets for decontamination falls somewhere between steam cleaning and abrasive blasting in effectiveness. Less effective than abrasive blasting, it has the advantage of producing liquid wastes (that can be processed) rather than solid wastes. As an external cleaning technique, it offers reduced airborne generation potential although this is offset by the need to control splashing. The utility of hydrolasing is generally limited to operations where internal accessibility is not required.

Ultrasonic Cleaning. Since this is an immersion process that is limited to smaller items, it is generally unsuitable for large scale decontamination. Although ultrasonic cleaning can be especially effective in removing contamination from crevices, it is doubtful that releasable levels can be reached consistently with this technique to make it a viable option.

Electropolishing. This is an electrochemical process where the object to be decontaminated serves as the anode in an electrolytic cell and radioactive contamination on the item is removed by anodic dissolution of the surface material. Although it is a relatively new process and has not yet been used for a full scale decontamination operation, it nevertheless requires consideration as a technique on the basis of its superior effectiveness in cleaning almost any metallic surface to a completely contamination-free state. On the other hand, this process has several limitations including the size of contaminated objects, the cost of electrolyte and special equipment, the consumption of considerable power, and the production of highly radioactive solutions.

Chemical Decontamination. Chemical flushing is recommended for remote decontamination of intact piping systems and their components. This technique uses concentrated or dilute solvents in contact with the contaminated item to dissolve either the contamination film covering the base metal or the base metal itself. Dissolution of the film is intended to be nondestructive to the base metal, and is generally used for operating facilities. Dissolution of the base metal, however, can be considered in a decommissioning program where reuse of the item will not occur.

Based on starting levels of contamination and required decontamination efforts, scrap metal inventories at nuclear power plants can be grouped into the following four categories:

- Low-level Surface Contaminated. This category is likely to consist of components that may be removed from previously specified buildings with significant residual radionuclide inventories but involve systems that are completely divorced from primary coolant, coolant waste streams, and other media with substantial levels of radioactivity. A sizeable fraction of scrap metal within this category will exhibit contamination that is limited to external surfaces and not exceed 100,000 dpm/100 cm^2. Decontamination strategies are likely to be routine with essentially 100% success at achieving scrap metal release limits.

- Medium-level Surface Contaminated. Metal components in direct contact with contaminated media that is below that of primary coolant and liquid radwaste may have internal and/or external surface contamination between 100,000 and 10,000,000 dpm/100 cm^2. Scrap metal in this category requires substantial decontamination efforts with less than 100% success in achieving unrestricted release.

- High-level Surface Contaminated. Scrap metal in this category will be represented by systems internally exposed to and contaminated by primary coolant and liquid radwastes leading to contamination levels in excess 10^7 dpm/100 cm^2. Reduced and variable fractions of metals are likely to be decontaminated to a level that permits unrestricted release.

- Volumetrically Contaminated. Components proximal to the reactor core may contain volumetrically-distributed activation products that range from nominal levels to extremely high levels. (Some of these components may also have high surface contamination.) With exception of melt-refining of such activated metals, removal of contaminants by standard processes is not achievable.

Table A5-1 identifies examples of major components that under normal operating conditions are likely to be grouped in the aforementioned categories. At a minimum, components that can be reasonably excluded from recycling include the reactor vessel and reactor vessel internals. For the Reference PWR, the reactor vessel is a right circular cylinder constructed of carbon steel about 216 mm in thickness and lined with 4 mm of stainless steel. The reactor vessel weight exclusive of internals is estimated at about 400 metric tons. The vessel internal structures support and constrain the fuel assemblies, direct coolant flow, guide in-core instrumentation, and provide some neutron shielding. Principal components include the lower core support assembly (inclusive of core barrel and shroud) and the upper core support and in-core instrumentation and support assemblies. These reactor vessel internals are made of Type 304 stainless steel and are estimated at about 190 metric tons.

The Reference BWR reactor vessel is also constructed of carbon steel with a stainless steel liner. The collective weight of the reactor vessel, top head, and internals, however, is estimated at 1,034 metric tons and is significantly higher than that of a PWR. Major reactor vessel internals include (1) the core shroud, (2) shroud support plate, (3) core support plate, (4) top fuel guide, (5) control rod guide tubes, (6) jet pumps, (7) shroud head and steam separator assembly, (8) steam dryer assembly, (9) feedwater spargers, and (10) core spray lines.

5.1 Identification of Contaminated Steel Components Suitable for Recycling

The elimination of scrap metal with significant levels of volumetrically-distributed activation products in carbon and stainless steel from recycling consideration yields Reference BWR and PWR systems/components defined in detail below (Tables A5-2A, B, C, D, and Tables A5-3A, B, C). These tables cite system components and their corresponding weights. The material composition of individual components have not been adequately defined. While a considerable number of components could be identified to consist exclusively of carbon steel or stainless steel, large quantities of steel exist as thick-walled carbon steel that is clad with thin-walled stainless steel (e.g., large piping, valves, vessels, tanks). When stainless steel provides corrosion resistant cladding, it is in effect physically inseparable from its large carbon steel component. In other instances, a given item will consist of many independent parts each having different material composition. For example, a recirculation pump may have a carbon steel casing and base with stainless steel shaft, impellers, and other internals. Although potentially separable, segregation of such individual parts is labor intensive and may be precluded by worker exposure (and ALARA) considerations and/or economic factors. A prudent approach may, therefore, assume that all steel scrap containing nickel be categorized as "stainless steel" (even if the nickel content is well below that of standard stainless steel alloys) because it is easier to upgrade scrap by adding nickel and other alloying material than it is to remove nickel for the production of mild steel or carbon steel.

Table A5-1. Examples of Scrap Metal Grouping Based on Contamination

Low-level Surface Contamination (on average < 100,000 dpm/100 cm^2): Minimal Effort to Decontaminate

- Structural metals in the turbine building, auxiliary building, and support buildings
- Control and instrumentation cables, cable trays
- Mechanical systems/piping not associated with primary coolant and radwastes

Medium Surface Contamination (on average between 1×10^5 and 1×10^7 dpm/100 cm^2): Substantial Effort Required to Decontaminate

- Containment spray recirculation
- Most auxiliary support systems
- BWR steam lines
- BWR turbines
- BWR condenser
- Containment building crane, refueling equipment, etc.
- Reactor building structural steel
- Fuel storage pool liner and water cleanup system

High Internal Surface Contamination (on average > 1×10^7dpm/100 cm^2): Aggressive Decontamination Required with <100% Success

- PWR primary recirculation piping
- PWR primary pumps and valves
- Liquid radwaste systems/tanks
- PWR steam generators
- Primary water cleanup system
- PWR pressurizer
- Coolant letdown and cleanup
- Spent fuel pool cooling

Significant Volumetric Contamination: Decontamination Unachievable

- Reactor vessel
- Reactor vessel top head
- Reactor vessel internals
- Control rod drive lines
- Reactor building components proximal to pressure vessel (< 10%)
- Rebar (~ 1% of plant total)

5.1.1 Reference BWR

For Reference BWR, a total of 29 contaminated systems is identified that are grouped by location (i.e., Reactor Building, Radwaste Building, and Turbine Building) (Table A5-2A, B, and C). Systems are cited in alphabetical order and identify the system-average level of contamination as previously defined in Table A5-1. Piping inventories for Reference BWR have been quantified and segregated by plant location in Table A5-2D.

In total, it is estimated that about 8.4 million kilograms or 8,400 metric tons of contaminated steel exist in Reference BWR that is potentially available for recycling. Based on material composition data cited in NUREG/CR-0672, it is further estimated that of the total 8,400 metric tons of steel, nearly 1,700 metric tons represents stainless steel. Stainless steel that is physically associated with carbon steel may, however, not be readily segregated.

Table A5-2A. Reference BWR Steel Inventories by Location
Within the Reactor Building

System: Containment Instrument Air
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
22	Instrument air accumulators	129	2,838
1	6" check valve	68	68
1	6" valve	82	82
222	Valves (3/4 - 2" dia.)	NA	4,008
TOTAL			6,996

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Control Rod Drive System
System Average Contamination Level: 80% Low; 20% Medium

Number	Component	Weight (kg)	
		Each	Total
460	CRD Blade	182	83,720
225	CRD Mechanism	218	49,050
185	Direction Control set	36	6,660
370	Scram valve	32	11,840
210	Scram accumulator	64	13,440
2	CRD Pump	1,816	3,632
2	Scram Discharge Volume	908	1,816
2	Pump Suction Filter	182	364
2	CRD drive water filter	45	90
2,660	Valves (3/4 - 4" dia.) & components	NA	48,830
TOTAL			219,442

System: Equipment Drain Processing
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Waste demineralizer	907	907
1	Waste collector filter	1,812	1,812
1	Waste filter hold pump	318	318
1	Waste collector tank & educator	10,229	10,229
1	Waste collector pump	284	284
1	Spent resin tank	657	657
1	Spent resin pump	102	102
1	Waste surge tank & educator	18,282	18,282
1	Waste surge pump	284	284
2	Waste sample tank & educator	6,960	13,920
2	Waste sample pump	230	462
199	Valves (1 - 8" dia.)	NA	5,374
TOTAL			52,631

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Fuel Pool Cooling and Cleanup
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
15	Spent Fuel Racks	18,424	276,360
1	Fuel Pool Liner	32,000	32,000
2	FPCC pumps	527	1,054
2	FPCC demin	1,566	3,132
2	Skimmer Surge Tank	5,354	10,708
2	FPCC Heat Exchanger	2,038	4,076
1	Supp. Pool Cleanup Pump	527	527
2	Resin Tank Agitator	36	72
1	Fuel Pool Precoat Pump	284	284
1	(Precoat) Dust Evacuator	104	104
2	FPCC hold pump	195	390
1	FPCC Precoat Tank	227	227
1	FPCC Resin Tank	227	227
165	Valves (1 - 10" dia.) & components	NA	8,038
TOTAL			337,199

System: High Pressure Core Spray
System Average Contamination Level: Medium

Number	Component	Material	Weight (kg)	
			Each	Total
2	24" Suction strainer		172	344
1	12 x 24" pump		27,410	27,410
1	1 x 2" pump		82	82
61	Valves (24 - 3/4" dia.)		NA	18,459
TOTAL				46,295

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: HVAC Components
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
7	Containment Recirc. Fans	636	4,452
5	Containment Fan Coil Units	1,500	7,500
17	Emergency Fan Foil Units	955	16,235
NA	Ducts (750 linear meters)	NA	29,975
TOTAL			58,162

System: Low Pressure Core Spray
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	24" Suction Strainer	172	344
1	Vent Strainer	43	43
1	14 x 24" pump	9,625	9,625
1	Pump pit	182	182
1	1 x 2" pump	82	82
45	Valves (3/4 - 24" dia.)	NA	10,523
TOTAL			20,799

System: Main Steam
System Average Contamination Level: 60% Medium; 40% Low

Number	Component	Weight (kg)	
		Each	Total
1	HP Turbine	194,169	194,169
2	LP Turbine	371,130	742,260
2	RFW Turbine	18,160	36,320
2	Steam Chest	55,565	111,130
1	Gland Steam Condenser	1,861	1,816
2	Ejector Condenser	1,816	3,632

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

Main Steam (Continued)

Number	Component	Weight (kg) Each	Weight (kg) Total
1	Moisture Separator	908	908
1	Bypass Valve Assy.	5,266	5,266
2	Moisture Separator Reheater	208,386	416,772
2	Steam Evaporator	13,472	26,944
4	2" Strainer	43	172
2	4" Strainer	100	200
2	12 Stop Check	894	1,788
4	30" Flow Restrictor	1,362	5,448
18	8" AO SRV	921	16,578
36	10" Vacuum Breakers	408	14,724
18	24 x 12" Quenchers	749	13,482
1	72" MOV	51,900	51,900
6	Stop Valves	18,160	108,960
6	Interceptor Valves	4,540	27,240
8	30" MSIV	636	5,088
1	24" MOV	3,223	3,223
4	24" Relief Valve	4,190	16,760
2	20" Relief Valve	3,496	6,992
1	16" MOV	1,920	1,920
2	16" Check Valve	1,534	3,068
2	14" Check Valve	1,008	2,016
2	14" MOV	1,253	2,506
2	12" MOV	1,135	2,270
8	28" HOV Governor Valves	3,632	29,056
951	Valves (1 - 10" dia.)	NA	69,592
TOTAL			1,922,200

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Main Steam Leakage Control
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
8	1/2" Valve	11	88
28	3/4" Valve	14	392
2	1" Flow Element	17	34
14	1" Valve	23	322
4	1" Check Valve	17	68
4	1-1/2" Flow Element	21	84
20	1-1/2" MOV	23	460
2	1-1/2" Check Valve	21	42
2	MSLC Fan (3")	204	408
4	MSLC Heater	57	227
TOTAL			2,125

System: Miscellaneous Items from Partial System
System Average Contamination Level: 55% Low; 45% Medium

Number	Component	Weight (kg)	
		Each	Total
5	TIP Drive Unit	361	1,805
2	TIP Indexing Unit	9	72
5	TIP Ball Valve	23	115
5	Explosive Shear Valve	23	115
5	TIP Shield Pig	154	770
1 set	TIP Tubing	295	295
2	Hogger (mechanical vacuum pump)	3,171	6,342
1	Refueling Bridge	24,918	24,918
1	Reactor Service Platform	5,210	5,210

Table A5-2A. Reference BWR Steel Inventory by Location Within the Reactor Building (Continued)

Miscellaneous Items from Partial System (Continued)

Number	Component	Weight (kg)	
		Each	Total
2	Refueling Mast	295	590
1	CRD Removal Turntable	2,492	2,492
1	CRD Removal Trolley	173	173
1	Incore Instrument Grapple	36	36
1	Fuel Support Piece Grapple	41	41
1	Control Blade Grapple	59	59
1	Spent Fuel Pool Work Table	445	445
2	Fuel Prep Machine	381	762
1	Channel Measurement Machine	422	422
185	Blade Guide	73	13,505
1	In Core Instrument Strongback	100	100
1	Manipulators, crows feet, etc.	136	136
20	In-vessel Manipulator Poles	14	280
9	Drywell Recirculation Fan	254	2,286
4	Stud Tensioner	1,044	4,176
1	RPV Head Strongback	2,134	2,134
1	Dryer/Separator Strongback	60	60
TOTAL			67,339

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Reactor Building, Closed Cooling Water
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
3	RBCCW Heat Exchanger	7,460	22,380
2	RBCCW Pump	1,597	3,194
1	RBCCW Surge Tank	531	531
5	Drywell Cooler & Fans	745	3,725
1	14" MOV	449	449
3	12" valve	331	993
7	10" MOV	250	1,750
6	10" Valve	250	1,500
4	10" Check Valve	168	672
1	10" Flow Element	16	672
218	Valves (3/4 - 8" dia.)	NA	6455
TOTAL			42,321

System: Reactor Building Equipment and Floor Drains
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
4	Drain Sump Pump	523	2,908
3	Drain Sump Pump	650	1,950
1	Equipment Drain Heat Exchanger	680	680
1	Drywell Equipment Drain HX	680	680
97	Valves (3/4 - 6" dia.)	NA	3,725
TOTAL			9,943

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Reactor Core Isolation Cooling
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Pelton Wheel Turbine/Pump	6,260	6,260
1	Barometric Condenser	553	553
1	Condenser Pump	679	679
1	Water Leg Pump	216	216
1	Vacuum Pump	453	453
1	Vacuum Tank	407	407
1	Steam Condensate Drip Pot	109	109
2	8" Suction Strainers	66	112
4	3/4" Steam Trap	25	100
1	10" Exhaust Drip Chamber	309	309
1	Turbine Exhaust Sparer	241	241
284	Valves (3/4 - 10" dia.)	NA	12,115
TOTAL			21,554

System: Reactor Water Cleanup System
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
2	RWCU Pump	590	1,180
2	Clean Up Hold Pump	534	1,068
1	Clean Up Precoat Pump	454	454
1	Sludge Discharge Pump	284	284
1	Decant Pump	102	102
2	Non-regenerative HX	4,086	8,172
3	Regenerative HX	4,131	12,394
2	Filter Demineralizer	3,178	6,356
1	Batch Tank	227	227
2	Phase Separator Tank	2,043	4,086
1	Precoat Agitator	27	27
259	Valves (1/2 - 6" dia.)	NA	13,170
TOTAL			47,520

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Residual Heat Removal
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
3	RHR Pump	7,792	23,376
1	Water Leg Pump	397	397
1	Drywell Upper Spray Ring Header	8,562	8,562
1	Drywell Lower Spray Ring Header	13,063	13,063
1	Wetwell Spray Ring Header	5,347	5,347
6	Suppression Pool Suction Strainers	195	1,171
2	RHR Heat Exchanger	29,190	58,380
3	24" MOV	7,150	21,450
2	20" MOV	4,086	8,172
1	20" Valve	4,086	4,086
11	18" MOV	4,603	50,633
8	18" Valve	4,603	36,828
5	18" Check	2,762	13,810
3	18" Flow Element	2,762	8,286
2	18" Restricting Orifice	2,762	5,524
4	16" MOV	2,724	10,896
4	14" MOV	1,544	6,176
2	14" Valve	1,544	3,088
3	14" Air Operated Check	971	2,913
2	14" Restricting Orifice	944	1,888
3	12" MOV	1,017	3,051
3	12" Valve	1,017	3,051
3	12" Air Operated Check	581	1,743
1	12" Restricting Orifice	549	549
2	10" Valve	731	1,462
1	10" Check Valve	399	399
324	Valves (3/4 - 3" dia.)	NA	12,100
TOTAL			306,401

Table A5-2A. Reference BWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Miscellaneous Drains
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Misc. Drain Tank #1	487	487
1	Misc. Drain Tank #2 w/ pumps	654	654
174	Valves (1" - 6" dia.)	NA	6,509
TOTAL			7,650

Table A5-2B. Reference BWR Steel Inventories for Locations Within the Radwaste Building

System: Chemical Waste Processing System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	Chemical Waste Tank	5,024	10,048
2	Detergent Drain Tank	1,834	3,668
2	Detergent Drain Pump	175	350
2	Concentrator Feed Pump	254	508
2	Chemical Waste Pump	478	956
1	Detergent Drain Filter	1,133	1,133
2	Chemical Addition Pump	257	454
2	Tank Agitators	36	72
2	Chemical Addition Pump	175	350
2	Distillate Tank	5,024	10,048
2	Distillate Tank Pump	230	460
1	Dist. Polishing Demineralizer	454	454
2	Decon Sol. Concentrator	3,405	6,810

Table A5-2B. Reference BWR Steel Inventories for Locations Within the Radwaste Building (Continued)

Chemical Waste Processing System (Continued)

Number	Component	Weight (kg)	
		Each	Total
2	Decon Sol. Concentrator Tank	711	1,422
2	Decon Conc. Recycle Pump	843	1,686
2	Decon Concentrator Condenser	2,305	4,610
2	Decon Concentrator Pre Heater	3,143	6,286
1	Decon Concentrator Waste Pump	254	508
2	Chemical Waste Stream Mixer	111	222
2	Condensate Receiver Tank	950	1,900
2	Condensate Receiver Tank Pump	102	204
293	Valves (1" - 8" dia.)	NA	7,654
TOTAL			59,803

System: Condensate Demineralizers
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
6	Filter Demineralizers	5,300	31,800
6	Resin Trap (w/ basket)	953	5,718
6	Demin Hold Pump	159	954
1	Condensate Backwash Rcving Tank	6,912	6,912
1	Sludge Disc Mixing Pump	420	420
1	Condensate Decant Pump	420	420
1	Condensate Backwash Transfer Pump	420	420
2	Condensate Phase Separator Tank	3,178	6,356
363	Valves & Components (1 - 36")	NA	36,783
TOTAL			89,783

Table A5-2B. Reference BWR Steel Inventories for Locations Within the Radwaste Building (Continued)

System: HVAC Components
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
11	RadWaste Air Handlers	1,327	14,597
3	Filter Units and Fans	11,123	33,369
NA	Ducts (1,980 linear meters)	NA	54,785
TOTAL			102,751

System: Radioactive Floor Drain Processing
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Floor Drain Demineralizer	907	907
1	Floor Drain Sample Tank	6,960	6,960
1	Floor Drain Sample Pump	230	230
1	Floor Drain Filter Aid Pump	118	118
1	Floor Drain Filter Hold Pump	317	317
1	Floor Drain Filter	1,812	1,812
1	Floor Drain Collector Pump	284	284
1	Floor Drain Collector Tank	10,229	10,229
1	Waste Decant Pump	102	102
1	Waste Sludge Dsch Mixing Pump	288	288
1	Waste Sludge Phase Sep Tank	5,490	5,490
171	Valves (1/2 - 8" dia.)	NA	4,500
TOTAL			31,237

Table A5-2B. Reference BWR Steel Inventories for Locations Within the Radwaste Building (Continued)

System: Rad Waste Building Drains
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
1	Chemical Drain Sump Pump	666	666
2	EDR Sump Pump	585	1,170
3	FDR Sump Pump	483	1,449
38	Valves & components (3/4 - 3" dia.)	NA	612
TOTAL			3,897

System: Standby Gas Treatment
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
42	2" Check Valve	25	1,050
2	18" Valves	2,225	4,450
14	18" Damper, MOV	563	7,882
2	18" Damper, AOV	563	1,126
2	SGT Filter Unit	8,898	17,796
8	3/4" Valve	14	112
4	Blower	2,043	8,172
TOTAL			40,588

Table A5-2C. Reference BWR Steel Inventories by Location Within the Turbine Building

System: Feed and Condensate
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	Turbine and Feed Pump	54,821	109,642
3	Condensate Booster Pump	12,006	36,018
3	Condensate Pump	21,883	65,649
1	Gland Exhaust Condenser	4,032	4,032
2	Air Ejector Condenser & Ejectors	6,614	13,228
1	Off Gas Condenser	897	897
2	#6 Feedwater Heater	73,394	146,788
2	#5 Feedwater Heater	68,863	137,726
3	#4 Feedwater Heater	35,338	106,014
3	#3 Feedwater Heater	50,288	150,864
3	#2 Feedwater Heater	51,194	153,582
3	#1 Feedwater Heater	62,974	188,922
2	Condensate Storage Tanks	50,475	100,950
2	Seal Steam Evaporator	13,451	26,902
2	Seal Steam Evap. Blowdown Cooler	213	426
407	Valves (1/2 - 24" dia.)	NA	350,478
TOTAL			1,592,118

Table A5-2C. Reference BWR Steel Inventories by Location Within the Turbine Building (Continued)

System: Extraction Steam
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
6	24" MOV	3,223	19,338
6	24" Stop Check	2,583	15,498
10	20" MOV	2,633	26,330
10	20" Stop Check	2,107	21,070
5	18" MOV	2,225	11,125
5	18" Stop Check	1,780	8,900
2	16" MOV	1,920	3,840
2	16" Stop Check	1,536	3,072
6	8" AOV	511	3,066
4	6" MOV	267	1,068
4	4" AOV	122	488
10	2" AOV	34	340
12	2" Restricting Orifice	25	300
85	Inst. root (typ 3/4" globe)	15	1,275
TOTAL			115,710

System: Heater Vents and Drains
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	Steam Evaporator Drain Tank	898	1,796
2	Heater Drain Tank	6,274	12,548
2	Moisture Separator Drain Tank	1,715	3,430
4	Reheater Drain Tank	1,134	4,536
4	Reheater Drain Tank	6,274	25,096
841	Valves & Components (1-1/2 - 20" dia.)	NA	151,369
TOTAL			198,775

Table A5-2C. Reference BWR Steel Inventories by Location Within the Turbine Building (Continued)

System: HVAC Components
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
4	Exhaust Air Units	4,900	19,600
1	Standby Gas Treatment	8,853	8,853
10	Air Handlers & Filter Units	829	8,290
NA	Ducts (1,000 linear meters)	NA	48,503
TOTAL			85,246

System: Offgas (Augmented) System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	Catalytic Recombiner Vessel	453	906
2	Preheater Heat Exchanger	538	1,076
1	Offgas Condenser	897	897
1	Water separator	271	271
2	Lab Vacuum Pump	45	90
2	Lab Vacuum Pump	45	90
2	Water Separator	1,359	1,718
8	Charcoal Ads. Vessel	4,077	32,615
2	Cooler Condenser	906	1,812
2	Pre-filter Vessel	1,133	2,266
2	After-filter Vessel	1,133	2,266
4	Desiccant Dryers	622	2,488
2	Dryer Heater	3,625	7,250
2	Dryer Chiller	2,265	4,530
2	Regen. Blower	636	1,272
9	6" Air Operated Valve	82	738
18	6" Valve	82	1,476
175	Valves (3/4 - 4" dia.)	NA	2,722
TOTAL			64,483

Table A5-2C. Reference BWR Steel Inventories by Location Within the Turbine Building (Continued)

System: Recirculation
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
2	Recirculation Pump w/motor	43,617	87,234
2	24" HOV	4,767	9,534
4	24" MOV	4,767	19,068
258	Valves (3/4 - 2" dia.)	NA	4,700
TOTAL			120,536

System: Turbine Building Drains
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
4	Equipment Drain Sump Pump	586	2,344
4	Floor Drain Sump Pump	484	1,936
25	Small Valves (2 - 3" dia.)	NA	450
TOTAL			4,730

Table 5-2D. Reference BWR Piping Inventories by Plant Location

Reactor Building
Average Contamination Level: Medium

Piping Material	Outside Diameter (mm)						
	<60	73 - 254	305 - 406	457 - 610	660 - 762	914 - 1,829	Total
Carbon Steel							
length (m)	2,323	3,922	505	952	55	---	
weight (kg)	8,479	110,368	61,897	127,160	14,850	---	322,754
Stainless Steel							
length (m)	6,169	500	54	---	---	---	
weight (kg)	18,674	4,551	2,143	---	---	---	25,368
Total Stock (kg)							348,122

Primary Containment
Average Contamination Level: High

Piping Material	Outside Diameter (mm)						
	<60	73 - 254	305 - 406	457 - 610	660 - 762	914 - 1,829	Total
Carbon Steel							
length (m)	263	1,084	211	1,239	374	559	
weight (kg)	1,366	63,181	29,760	554,877	145,312	234,882	1,029,378
Stainless Steel							
length (m)	3,850	110	64	55	---	---	
weight (kg)	10,603	3,411	8,789	21,440	---	---	44,243
Total Stock (kg)							1,073,621

Table 5-2D. Reference BWR Piping Inventories by Plant Location (Continued)

Turbine Building
Average Contamination Level: Low

Piping Material	Outside Diameter (mm)						
	<60	73 - 254	305 - 406	457 - 610	660 - 762	914 - 1,829	Total
Carbon Steel							
length (m)	3,336	2,632	1,647	1,832	465	559	
weight (kg)	14,153	115,525	176,600	386,321	240,698	234,882	1,168,179
Stainless Steel							
length (m)	---	38	103	---	---	---	
weight (kg)	---	1,474	6,421	---	---	---	7,895
Total Stock (kg)							1,176,074

RadWaste and Control Buildings
Average Contamination Level: High

Piping Material	Outside Diameter (mm)						
	<60	73 - 254	305 - 406	457 - 610	660 - 762	914 - 1,829	Total
Carbon Steel							
length (m)	3,087	3,337	338	12	---	99	
weight (kg)	10,267	75,778	29,221	4,584	---	29,410	149,260
Stainless Steel							
length (m)	1,150	1,026	55	---	---	---	
weight (kg)	4,747	10,164	1,756	---	---	---	16,667
Total Stock (kg)							165,927

5.1.2 Reference PWR

Data in Table A5-3A, B, and C identify major contaminated PWR components by function and location. The total inventory of steel (excluding the reactor pressure vessel and its internals) is estimated at about 4,100 metric tons. It should be pointed out, however, that slightly less than half or about 2,000 metric tons are contributed by primary system components that include steam generators, pressurizer, reactor coolant piping, etc. (Table A5-3A). The long-term buildup of activated corrosion products and leaked fission products on internal surfaces among these components are projected to be high. Even with intense and aggressive decontamination efforts, their release for unrestricted recycling may not be technically achievable or may be precluded by radiological and other concerns.

The balance of about 2,100 metric tons includes eleven (11) internally contaminated reactor support systems and piping that are associated with the Auxiliary Building/Fuel Storage facility and a variety of structural components where contamination is limited to external surfaces. It is estimates that nearly 20% of all steel is stainless steel.

Table A5-3A. Reference PWR Contaminated Steel Inventories by Location Within the Reactor Building

System: External Surface Structures Equipment
System Average Contamination Level: 70% Low; 30% Medium

Number	Component	Weight (kg)	
		Each	Total
NA	Refueling Cavity Liner	17,000	17,000
NA	Base Liner	54,000	54,000
NA	Reactor Cavity Liner	14,500	14,500
NA	Floor and Cavity Liner Plates	139,000	139,000
NA	CRD Missile Shield	11,000	11,000
NA	Stairways/Gratings	45,000	45,000
NA	Miscellaneous Equipment	NA	13,600
TOTAL			294,100

Table A5-3A. Reference PWR Steel Inventories by Location Within the Reactor Building (Continued)

System: Internally Contaminated Primary System Components
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
4	Steam Generator	312,000	1,248,000
4	Rx Coolant Pumps	85,350	341,400
1	Pressurizer	88,530	88,530
NA	Containment Spray Piping		90,800
1	Pressurizer Relief Tank	12,338	12,338
4	Safety Inject. Syst. Accumulator	34,700	138,800
1	Reactor Cavity Drain Pump	363	363
2	Containment Sump Pump	635	1,270
1	Excess Letdown Heat Exchanger	726	726
1	Regenerative Heat Exchanger	2,994	2,994
NA	<u>Reactor Coolant Piping Data</u>		
	Size: 686 - 787 mm ID/Length 81 m		100,698
	Size: 51 - 356 mm OD/Length 677 m		11,793
TOTAL			2,037,712

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage

System: Component Cooling Water System
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
2	CCW Heat Exchanger	31,780	63,560
2	CCW Pump	6,810	13,620
1	CCW Surge Tank	908	1,816
1	Chem. Addition Tank	477	954
9	Sample Heat Exchanger	3,178	28,602
169	Valves (3/4 - 24" dia.)		104,700
TOTAL			213,252

System: Containment Spray System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
2	Pump	3,087	6,174
2	Pump	45	90
1	Tank	2,490	2,490
6	Small Electrical Equipment	34	204
6	Large Electrical Equipment	68	408
46	Valves (3/4 - 18" dia.)	NA	37,875
TOTAL			47,241

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Clean Radioactive Waste Treatment System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Rx Coolant Drain Tank	758	758
2	Rx Coolant Drain Pump	227	454
1	Rx Coolant Drain Filter	159	159
1	Spent Resin Storage Tank	3,087	3,087
2	Clean Waste Recv. Tank	4,975	9,950
2	Clean Waste Recv. Pump	227	454
2	Treated Waste Mon. Tank	5,085	10,170
2	Treated Waste Mon. Pump	104	208
1	Aux. Building Drain Tank	949	949
2	Aux. Building Drain Pump	590	1,180
1	Chem. Waste Drain Tank	2,452	2,452
2	Chem. Waste Drain Pump	91	182
1	Waste Conc. Hold Tank	949	949
1	Waste Conc. Hold Pump	104	104
1	Clean Waste Filter	30	30
1	Clean Radwaste Evaporator	18,160	18,160
83	Valves (2 - 3" dia.)	NA	3,935
TOTAL			53,181

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Control Rod Drive System
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
4	Small Electric Equipment	34	136
4	Large Electric Equipment	68	272
1	Large Mech. Equipment	68	68
TOTAL			476

System: Electrical Components and Annunciators
System Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
2	125 VDC Power (Small)	68	136
2	125 VDC Power (Medium)	227	454
1	125 VDC Power (Large)	2,270	2,270
1	4.16 KV AC & Aux. (Small)	227	227
1	4.16 KV AC & Aux. (Large)	9,080	9,080
7	480 KV AC Ld Cntr (Small)	227	1,589
7	480 KV AC Ld Cntr (Large)	908	6,356
1	480 KV AC MCC	227	227
12	480 KV AC MCC	9,080	108,960
2	Annunciators (elec. port.)	34	68
22	Annunciators (mech. port.)	34	748
TOTAL			130,115

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Chemical and Volume Control System
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
3	Regenerative Heat Exchanger	2,724	8,172
1	Seal Water Heat Exchanger	772	772
1	Letdown Heat Exchanger	863	863
1	Excess Letdown Heat Exchanger	726	726
2	Centrif. Charge Pump	7,759	15,518
1	Volume Control Tank	2,202	2,202
3	Holdup Tank	13,620	40,860
2	Monitor Tank	9,080	18,160
2	Boric Acid Tank	9,080	18,160
1	Batch Tank	658	658
1	Resin Fill Tank	118	118
1	Reciprocal Charge Pump	8,036	8,036
2	Boric Acid Pump	281	562
1	Reactor Coolant Filter	91	91
2	Mixed Bed Demineralizer	477	954
1	Cation Ion Exchange	477	477
2	Seal Injection Filter	749	1,498
1	Concentrate Hold Tank	1,589	1,589
3	Evaporator Feed Ion Exchange	477	1,431
2	Evaporator Condensate Ion Exchange	477	954
2	Condensate Filter	18	18
1	Concentrates Filter	18	18
1	Conc. Hold Tank Transfer Pump	91	182
2	Gas Stripper Feed Pump	227	454
2	Boric Acid Evap. Skid Assembly	9,489	18,978
1	Ion Exchange Filter	68	68
1	Recirculation Pump	288	288
378	Valves (3/4 - 6" dia.)	NA	17,481
TOTAL			159,288

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Dirty Radioactive Waste Treatment System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Rx Cavity Drain Pump	363	363
2	Rx Cont. Sump Pump	681	1,362
1	Dirty Waste Monitor Tank	2,633	2,633
2	Dirty Waste Monitor Tank Pump	91	182
1	Dirty Waste Drain Tank	2,969	2,969
2	Dirty Waste Drain Tank Pump	181	362
2	Aux Building Sump Pump	590	1,180
46	Valves (2 -3" dia.)	NA	2,280
TOTAL			11,331

System: Radioactive Gaseous Waste System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
1	Surge Tank	404	404
4	Decay Tank	4,900	19,600
2	Gas Compressor	3,632	7,264
2	Moisture Separator	45	90
2	HEPA/prefilter	91	182
1	Exhaust Fan	45	45
2	Br. Seal Water Heat Exchanger	3,496	6,992
4	Large Electrical Equipment	68	272
2	Large Mechanical Equipment	2,270	4,540
1	HVAC Equipment	68	68
83	Valves (3/4 - 4" dia.)	NA	4,607
TOTAL			44,064

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Residual Heat Removal System
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
2	Pump	3,087	6,174
2	Heat Exchanger Unit	10,487	20,974
12	Small Electrical Equipment	34	408
11	Large Electrical Equipment	68	748
1	Small Mechanical Equipment	34	34
42	Valves (3/8 - 14" dia.)	NA	49,032
TOTAL			77,370

System: Safety Injection System
System Average Contamination Level: Medium

Number	Component	Weight (kg)	
		Each	Total
4	Accumul. Tank	34,731	138,924
1	B. Inj. Tank	12,939	12,939
2	Safety Inj. Pump	3,904	7,808
1	Refueling Water Tank	80,721	80,721
1	Primary Water Storage Tank	45,037	45,037
10	Small Electrical Equipment	34	340
10	Large Electrical Equipment	68	680
1	Small Mechanical Equipment	34	34
89	Valves (3/4 - 10" dia.)	NA	12,114
TOTAL			298,597

Table A5-3B. Reference PWR Steel Inventories by Location Within the Auxiliary Building and Fuel Storage (Continued)

System: Spent Fuel System
System Average Contamination Level: High

Number	Component	Weight (kg)	
		Each	Total
1	Pump	454	454
2	Pump	409	918
1	Pump	318	318
2	Filter	163	326
1	Filter	68	68
1	Demineralizer	998	998
2	Heat Exchanger	2,769	5,538
53	Valves (3/4 - 10" dia.)	NA	14,117
1	Fuel Pool Liner	37,000	37,000
	Fuel Storage Racks		49,079
	Fuel Handling System		18,470
	Overhead Crane		113,000
TOTAL			240,286

Structural Steel Components
Average Contamination Level: Low

Number	Component	Weight (kg)	
		Each	Total
NA	Wall Support	NA	24,200
NA	Roof Support	NA	16,300
NA	Stairs/Grates/Tracks/Hand-rails	NA	33,200
NA	I-beams	NA	207,000
NA	HVAC Ducts	NA	26,550
NA	HVAC Components	NA	76,500
TOTAL			383,750

Table A5-3C. Reference PWR Non-RCS Stainless Steel Piping[a] [b]

Nominal Size, In.	Schedule	Lineal Meters	Total Inside Surface Area (m^2)	Total Weight (kg)
1/2	80	122	5.3117	198
	160	122	4.5140	238
3/4	40	122	8.0163	205
	80	183	10.8278	400
	160	580	28.2806	1,671
1	40	61	5.1026	152
	80	61	13.9652	590
	160	427	22.7504	1,803
1-1/2	40	122	15.6628	493
	80	335	40.1299	1,810
	160	549	58.5751	3,967
2	40	305	50.2718	1,655
	80	488	75.4540	3,642
	160	1,067	143.6043	11,840
3	160	140	29.3566	2,985
4	160	183	50.1697	6,128
6	160	311	128.6770	20,972
8	160	143	70.6775	15,923
10	140	192	134.0704	29,750
12	140	88	74.0579	18,370
14	140	100	92.2988	24,474
TOTAL			1,061.7644	147,266

[a] Includes piping for the following systems: Residual Heat Removal, Chem and Volume Control, Emergency Core Cooling, Containment Spray, Auxiliary Feedwater, Spent Fuel Pool Cooling, Condensate Facility, Station Service, Component Cooling, Service Cooling, Makeup Water System.

[b] Contamination levels are likely to vary over several orders of magnitude from near background levels up to as high as 1 x 10^7 dpm/100 cm^2. About 80% will be considered low-level contaminated with the remaining 20% at medium-level.

5.1.3 Summary Estimates of Steel for Reference BWR/PWR and the Commercial Nuclear Industry

Table A5-4 presents summary data for contaminated steel potentially available for recycling. Estimates for the Reference BWR and PWR were derived by summing component mass values previously cited in Table A5-2 and Table A5-3, respectively. Fractional quantities of stainless steel were developed from information provided by Bryan and Dudley (1974) as well as NUREG/CR-0672 and NUREG/CR-0130. For example, Table A4-2 in Section 4.1 of this report cited a total stainless steel quantity used in the construction of a Reference PWR. Based on PWR design parameters, it was concluded that all of the 1,154.6 metric tons of stainless steel in the Reactor Plant Equipment and 21.1 metric tons for Fuel Storage were contaminated for a total of about 1,175 metric tons. Included in this total, however, was about 348 metric tons of stainless steel that is volumetrically contaminated with activation products that are considered unsuitable for recycling. Thus, Table A5-4 cites a net mass of 827 metric tons of stainless steel, which, when subtracted from the total mass of 4,138 metric tons of contaminated steel, leaves the remainder of 3,311 metric tons of contaminated carbon steel.

Estimates for the entire commercial nuclear industry were derived by taking Reference BWR and Reference PWR values and applying previously cited plant-specific scaling factors for each of the 40 BWRs and 83 PWRs (see Addendum #1). Approximately 600,000 metric tons of contaminated steel over time may become available for recycling. About 80% of the contaminated steel is carbon steel with stainless steel representing the balance.

Because past and current regulatory release criteria (i.e., U.S. NRC Regulatory Guide 1.86) are defined in activity levels per unit surface area, information cited in this section has been presented in this fashion. However, for risk analysis pertaining to recycling of contaminated scrap metals, a more meaningful approach is to express contamination levels in terms of activity per unit mass. This conversion required the derivation of the average mass thickness (g/cm^2) of contaminated metal surfaces by the following equation:

$$\textit{Average Mass Thickness } (g/cm^2) = \frac{\sum \textit{Contaminated Surface Areas } (cm^2)}{\sum \textit{Contaminated Metal Mass } (g)}$$

For the contaminated systems/components previously identified for Reference BWR and PWR, a weighted average mass density of 3.5 g/cm^2 for contaminated surfaces was estimated. At a density of about 8 g/cm^3 for steel, this corresponds to an average thickness of about 4.4 mm (0.17 inches). This average mass thickness can now be readily applied to estimate the activity level per unit mass of contaminated steel. For example, under the current interim release criteria of 5,000 dpm/100 cm^2 for beta-gamma emitters, the residual contamination on average would correspond to about 14 dpm/g (or about 6.5 pCi/g.; or 0.23 Bq/g) of steel.

Table A5-4. Summary Data for Contaminated Steel Inventories Potentially Suitable for Recycling

Contaminated Material[a]	Quantity (metric tons)				
	Reference BWR[b]	Reference PWR[c]	All BWRs[b]	All PWRs[c]	Total Industry
Stainless Steel	1,688	827	56,987	64,738	121,725
• Low-level Contamination (<1 x 10^5 dpm/100 cm^2)	576	210	19,446	16,439	35,885
• Medium-level Contamination (1 x 10^5 to 1 x 10^7 dpm/100 cm^2)	786	114	26,535	8,924	35,459
• High-level Contamination (>1 x 10^7 dpm/100 cm^2)	326	503	11,006	39,375	50,381
Carbon Steel	6,754	3,311	228,015	259,185	487,200
• Low-level Contamination (<1 x 10^5 dpm/100 cm^2)	2,306	841	77,851	65,834	143,685
• Medium-level Contamination (1 x 10^5 to 1 x 10^7 dpm/100 cm^2)	3,146	458	106,209	35,852	142,061
• High-level Contamination (>1 x 10^7 dpm/100 cm^2)	1,302	2,012	43,955	157,499	201,454
TOTALS	8,442	4,138	285,002	323,923	608,925

[a] Although data for stainless steel and carbon steel are presented as independent quantities, it must be acknowledged that a significant fraction of stainless steel is unlikely to be segregated as such for recycling purposes.

[b] BWR radionuclide contaminants reflect those of Table A3-11 and Table A3-21.

[c] PWR radionuclide contaminants reflect those of Table A3-17 and Table A3-20.

5.2 Metal Inventories Other Than Steel

Although steel is clearly the dominant metal used in the construction and system components that define a nuclear power plant, there are also significant quantities of other metals that may be suitable for recycling. Tables A4-1 and A4-3 in Section 4 of this report had identified total plant inventories for Reference BWR and Reference PWR for the following metals and metal alloys: (1) galvanized iron, (2) copper, (3) inconel, (4) lead, (5) bronze, (6) aluminum, (7) brass, (8) nickel, and (9) silver. However, there exist no credible data in the open literature regarding the estimated fraction(s) of these metal inventories that are likely to be contaminated or the extent of their contamination.

In the absence of reported data, a reasonable approach may assume that the contaminated fraction among total plant inventories of these metals parallels the contaminated fraction of carbon steel for Reference BWR and Reference PWR. Justification for this modeling approach is based on the fact that most of these metals exist as sub-components of larger items consisting primarily of carbon steel. From data cited in previous sections, the percent of contaminated carbon steel suitable for recycling to that of total plant inventory corresponds to 20% and 10% for Reference BWR and Reference PWR, respectively. The application of these values yields contaminated metal quantities (suitable for recycling) cited in Table A5-5. Due to physical differences and chemical properties that affect corrosion and internal contamination, a parallel approach to quantify metal inventories as low-level, medium-level, and high-level contaminated seems inappropriate.

Table A5-5. Summary of Metal Quantities Other than Steel
(metric tons)

Metal Type	Reference Facility		Industry		
	BWR	PWR	All BWRs	All PWRs	Total
Galvanized Iron	258	130	8,710	10,037	18,747
Copper	137	69	4,625	5,327	9,952
Inconel	24	12	810	927	1,737
Lead	9.1	4.6	307	355	662
Bronze	5.0	2.5	169	193	362
Aluminum	3.6	1.8	122	139	261
Brass	2.0	1.0	68	77	145
Nickel	0.2	0.1	7	8	15
Silver	<0.2	<0.1	<7	<8	<15

5.3 Time-Table for the Availability of Scrap Metal from the Decommissioning of Nuclear Power Plants

For currently operating nuclear power plants, an operational period of 40 years is assumed. The projected year of shutdown for each of 123 reactor units is identified in Addendum #1. Following reactor shutdown, a minimum of 10 years is assumed before significant dismantling activity could proceed that yields significant quantities of scrap metal. Thus, for currently operating reactors, the earliest dates for releasing scrap metal are defined by their startup dates plus 50 years. Currently, there are 8 reactor units that have been permanently shutdown (Dresden-1 (1984); Indian Point-1 (1980); LaCrosse (1981); TMI-2 (1979); Humboldt Bay (1976); Trojan (1993); Rancho Seco (1989); San Onofre-1 (1992); and Yankee Rowe (1992)). A conservative assumption for these facilities projects the release of scrap metal over a ten-year period between 2000 and 2009.

Table A5-6 summarizes the availability of scrap for yearly intervals starting with the year 2010. The incremental quantity of scrap metal available for recycling is illustrated in Figure A5-1. The release of scrap metal based on this time-table must, however, be considered highly conservative since many, if not most, facilities are likely to delay D&D activities for varying portions of the allowable 50-year SAFSTOR period.

Table A5-6. Time-Table for Available Scrap Metals from Decommissioned Nuclear Power Plants
Quantities (metric tons)

Year	CS	SS	Galv. Iron	Copper	Inconel	Lead	Bronze	Alum.	Brass	Nickel
2000-2009	15,377	4,107	609	323	57	21	12	8	5	0.5
2010	804	105	31	17	3	1	<1	<1	<1	—
2011	—	—	—	—	—	—	—	—	—	—
2012	—	—	—	—	—	—	—	—	—	—
2013	—	—	—	—	—	—	—	—	—	—
2014	3,616	475	140	75	13	5	3	2	1	—
2015	3,616	475	140	75	13	5	3	2	1	—
2016	—	—	—	—	—	—	—	—	—	—
2017	6,464	1,958	257	136	24	9	5	4	2	0.2
2018	14,811	2,712	580	308	54	20	11	8	4	0.5
2019	1,763	534	70	37	7	2	1	1	<1	0.1

Table A5-6. Time-Table for Available Scrap Metals from Decommissioned Nuclear Power Plants (Continued)
Quantities (metric tons)

Year	CS	SS	Galv. Iron	Copper	Inconel	Lead	Bronze	Alum.	Brass	Nickel
2020	15,442	2,740	604	321	56	21	12	8	4	0.5
2021	2,438	739	97	51	9	3	2	1	<1	0.1
2022	10,328	1,674	370	197	34	13	7	5	3	0.3
2023	38,415	8,571	1,512	439	141	53	29	21	12	1.2
2024	41,117	8,855	1,616	859	151	57	31	22	12	1.3
2025	12,927	3,175	510	271	48	18	10	7	4	0.4
2026	22,927	5,326	903	480	84	31	17	12	7	0.7
2027	5,611	1,700	223	118	21	8	4	3	2	0.2
2028	9,574	2,185	377	200	35	13	7	5	3	0.3
2029	—	—	—	—	—	—	—	—	—	—
2030	9,078	2,750	362	192	34	13	7	5	3	0.3
2031	8,961	2,714	357	189	34	12	7	5	3	0.3
2032	10,697	1,406	415	221	38	15	8	6	3	0.3
2033	11,191	2,468	440	234	41	16	8	6	3	0.3
2034	30,466	5,672	1,193	635	111	42	23	16	9	1.0
2035	26,173	6,206	1,032	548	96	36	20	14	8	0.8
2036	32,396	6,248	1,269	675	118	45	24	17	10	1.0
2037	13,192	3,996	525	278	49	18	10	7	4	0.4
2038	9,637	2,919	384	203	36	13	7	5	3	0.3
2039	8,366	1,639	328	174	30	12	6	4	2	0.2
2040	12,956	3,925	516	273	48	18	10	7	4	0.4
2041	—	—	—	—	—	—	—	—	—	—
2042	3,261	988	130	69	12	5	2	2	1	0.1
2043	—	—	—	—	—	—	—	—	—	—
2044	—	—	—	—	—	—	—	—	—	—
2045	2,703	819	107	57	10	4	2	1	1	0.1
2046+	12,868	3,902	512	271	48	18	10	7	4	0.4
Total	397,175	90,983	15,609	7,926	1,455	547	299	212	121	12.2

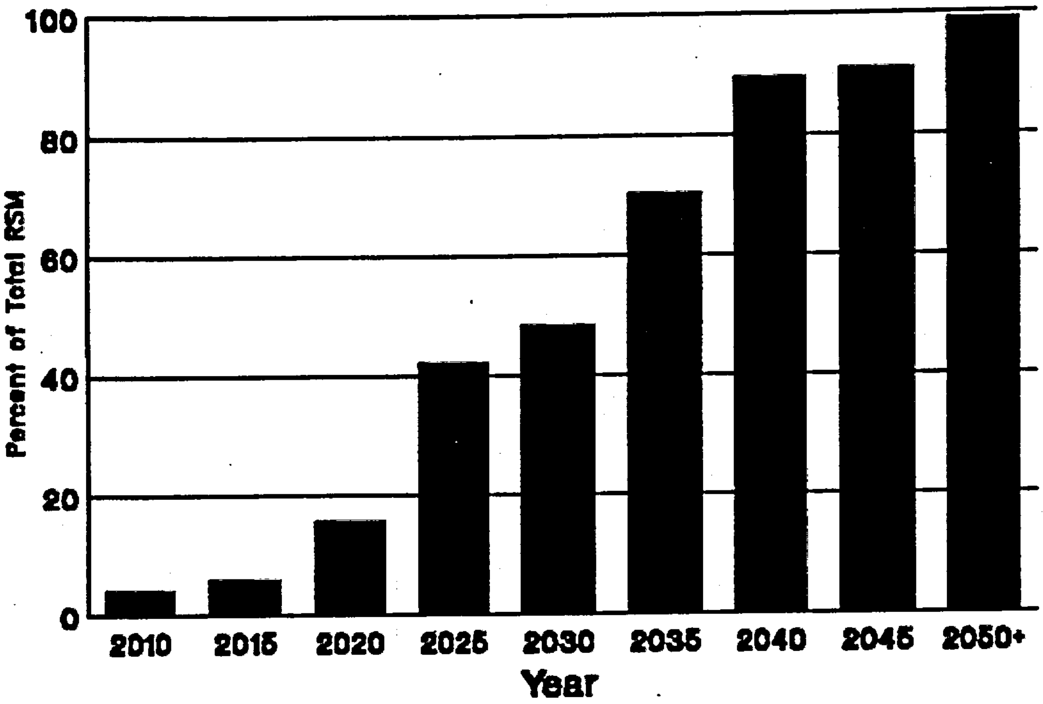

Figure A5-1. Cumulative Availability of Scrap Metal from Nuclear Utilities

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ADDENDUM #1

Demographic Data for the U.S. Nuclear Power Industry

Electric Utility Name	Reactor Name	Reactor Type	Power Rating (MWe)	Scaling Factor*	Projected Year of Shutdown
Arizona Power Company	Farley 1	PWR (West.)	812	0.87	2017
Arizona Power Company	Farley 2	PWR (West.)	824	0.88	2021
Arizona Public Service	Palo Verde 1	PWR (CE)	1,270	1.18	2024
Arizona Public Service	Palo Verde 2	PWR (CE)	1,270	1.18	2025
Arizona Public Service	Palo Verde 3	PWR (CE)	1,270	1.18	2027
Arkansas Power & Light	Arkansas Nuc. 1	PWR (B&W)	836	0.89	2014
Arkansas Power & Light	Arkansas Nuc. 2	PWR (CE)	858	0.90	2018
Baltimore Gas & Electric	Calvert Cliffs 1	PWR (CE)	825	0.88	2014
Baltimore Gas & Electric	Calvert Cliffs 2	PWR (CE)	825	0.88	2016
Boston Edison	Pilgrim 1	BWR (GE)	663	0.76	2012
Carolina Power & Light	Brunswick 1	BWR (GE)	767	0.84	2016
Carolina Power & Light	Brunswick 2	BWR (GE)	754	0.83	2014
Carolina Power & Light	Harris 1	PWR (West.)	860	0.91	2026
Carolina Power & Light	Robinson 2	PWR (West.)	683	0.78	2010
Cleveland Electric	Perry 1	BWR (GE)	1,167	1.11	2026
Commonwealth Edison	Braidwood 1	PWR (West.)	1,090	1.06	2028
Commonwealth Edison	Braidwood 2	PWR (West.)	1,090	1.06	2028
Commonwealth Edison	Byron 1	PWR (West.)	1,120	1.08	2025
Commonwealth Edison	Byron 2	PWR (West.)	1,120	1.08	2027
Commonwealth Edison	Dresden 1	BWR (GE)	200	0.34	1984
Commonwealth Edison	Dresden 2	BWR (GE)	772	0.84	2010
Commonwealth Edison	Dresden 3	BWR (GE)	773	0.84	2013
Commonwealth Edison	LaSalle County 1	BWR (GE)	1,048	1.03	2024
Commonwealth Edison	LaSalle County 2	BWR (GE)	1,048	1.03	2024
Commonwealth Edison	Quad Cities 1	BWR (GE)	769	0.84	2013
Commonwealth Edison	Quad Cities 2	BWR (GE)	769	0.84	2013

Electric Utility Name	Reactor Name	Reactor Type	Power Rating (MWe)	Scaling Factor*	Projected Year of Shutdown
Commonwealth Edison	Zion 1	PWR (West.)	1,040	1.03	2013
Commonwealth Edison	Zion 2	PWR (West.)	1,040	1.03	2014
Consolidated Edison	Indian Point 1	PWR (B&W)	265	0.41	1980
Consolidated Edison	Indian Point 2	PWR (West.)	931	0.96	2013
Consumers Power	Big Rock Point	BWR (GE)	67	0.16	2000
Consumers Power	Palisades	PWR (CE)	755	0.83	2011
Dairyland Power Co-op	LaCrosse	BWR (AC)	50	0.14	1981
Detroit Edison	Enrico Fermi 2	BWR (GE)	1,060	1.04	2025
Duke Power	Catawba 1	PWR (West.)	1,129	1.09	2025
Duke Power	Catawba 2	PWR (West.)	1,129	1.09	2026
Duke Power	McGuire 1	PWR (West.)	1,129	1.09	2021
Duke Power	McGuire 2	PWR(West.)	1,129	1.09	2023
Duke Power	Oconee 1	PWR (B&W)	846	0.90	2013
Duke Power	Oconee 2	PWR (B&W)	846	0.90	2013
Duke Power	Oconee 3	PWR (B&W)	846	0.90	2014
Duquesne Light	Beaver Valley 1	PWR (West.)	810	0.87	2016
Duquesne Light	Beaver Valley 2	PWR (West.)	833	0.89	2026
Florida Power Corp.	Crystal River 3	PWR (B&W)	820	0.88	2016
Florida Power & Light	St. Lucie 1	PWR (CE)	839	0.89	2016
Florida Power & Light	St. Lucie 2	PWR (CE)	839	0.89	2023
Florida Power & Light	Turkey Point 3	PWR (West.)	666	0.76	2007
Florida Power & Light	Turkey Point 4	PWR (West.)	666	0.76	2007
Georgia Power	Hatch 1	BWR (GE)	744	0.82	2014
Georgia Power	Hatch 2	BWR (GE)	762	0.83	2018
Georgia Power	Votle 1	PWR (West.)	1,105	1.07	2027
Georgia Power	Votle 2	PWR (West.)	1,103	1.07	2029
GPU Nuclear	Oyster Creek	BWR (GE)	610	0.72	2004
GPU Nuclear	Three Mile Island 1	PWR (B&W)	808	0.87	2014

Electric Utility Name	Reactor Name	Reactor Type	Power Rating (MWe)	Scaling Factor*	Projected Year of Shutdown
GPU Nuclear	Three Mile Island 2`	PWR (B&W)	808	0.87	1979
Gulf States Utilities	River Bend 1	BWR (GE)	936	0.96	2025
Houston Lighting & Power	South Texas 1	PWR (West.)	1,250	1.16	2027
Houston Lighting & Power	South Texas 2	PWR (West.)	1,250	1.16	2028
Illinois Power	Clinton 1	BWR (GE)	930	0.95	2026
Indiana Michigan Power	Cook 1	PWR (West.)	1,000	1.00	2014
Indiana Michigan Power	Cook 2	PWR (West.)	1,060	1.04	2017
Iowa Electric Light	Duane Arnold	BWR (GE)	515	0.64	2014
Kansas Gas & Electric	Wolf Creek 1	PWR (West.)	1,131	1.09	2025
Louisiana Power & Light	Waterford 3	PWR (CE)	1,075	1.05	2024
Maine Yankee Atomic Power	Maine Yankee	PWR (CE)	870	0.91	2008
Nebraska Public Power	Cooper Station	BWR (GE)	778	0.85	2014
New York Power Authority	Fitzpatrick	BWR (GE)	800	0.86	2015
New York Power Authority	Indian Point 3	PWR (West.)	980	0.99	2015
Niagara Mohawk	Nine Mile Point 1	BWR (GE)	605	0.72	2005
Niagara Mohawk	Nine Mile Point 2	BWR (GE)	1,080	1.06	2026
Northeast Utilities	Haddam Neck	PWR (West.)	560	0.68	2007
Northeast Utilities	Millstone 1	BWR (GE)	652	0.75	2010
Northeast Utilities	Millstone 2	PWR (CE)	863	0.91	2015
Northeaast Utilities	Millstone 3	PWR (West.)	1,137	1.09	2025
Northern States Power	Monticello	BWR (GE)	532	0.66	2010
Northern States Power	Prairie Island 1	PWR (West.)	507	0.64	2013
Northern States Power	Prairie Island 2	PWR (West.)	503	0.63	2014
Omaha Public Power	Fort Calhoun	PWR (CE)	476	0.61	2008
Pacific Gas & Electric	Diablo Canyon 1	PWR (West.)	1,073	1.05	2025
Pacific Gas & Electric	Diablo Canyon 2	PWR (West.)	1,087	1.06	2026
Pacific Gas & Electric	Humbolt Bay	BWR (GE)	65	0.16	1976

Electric Utility Name	Reactor Name	Reactor Type	Power Rating (MWe)	Scaling Factor*	Projected Year of Shutdown
Pennsylvania Power	Susquehanna 1	BWR (GE)	1,040	1.03	2022
Pennsylvania Power	Susquehanna 2	BWR (GE)	1,044	1.03	2024
Philadephia Electric	Limerick 1	BWR (GE)	1,055	1.04	2024
Philadelphia Electric	Limerick 2	BWR (GE)	1,055	1.04	2029
Philadelphia Electric	Peach Bottom 2	BWR (GE)	1,051	1.04	2008
Philadelphia Electric	Peach Bottom 3	BWR (GE)	1,035	1.02	2008
Portland General Electric	Trojan	PWR (West.)	1,104	1.07	1993
Public Service E & G	Hope Creek	BWR (GE)	1,031	1.02	2026
Public Service E & G	Salem 1	PWR (West.)	1,106	1.07	2016
Public Service E & G	Salem 2	PWR (West.)	1,106	1.07	2020
Public Service of NH	Seabrook	PWR (West.)	1,150	1.10	2030
Rochester Gas & Electric	Ginna	PWR (West.)	470	0.60	2009
Sacramento Municipal	Rancho Seco	PWR (B&W)	873	0.91	1989
South Carolina E & G	Summer	PWR (West.)	885	0.92	2035
Southern California Ed.	San Onafre 1	PWR (West.)	436	0.58	1992
Southern California Ed.	San Onafre 2	PWR (CE)	1,070	1.05	2013
Southern California Ed.	San Onafre 3	PWR (CE)	1,080	1.06	2013
Systems Energy Resources	Grand Gulf 1	BWR (GE)	1,143	1.10	2022
Tennessee Valley Authority	Bellefonte 1	PWR (B&W)	1,235	1.15	2038
Tennessee Valley Authority	Bellefonte 2	PWR (B&W)	1,235	1.15	2043
Tennessee Valley Authority	Browns Ferry 1	BWR (GE)	1,065	1.04	2013
Tennessee Valley Authority	Browns Ferry 2	BWR (GE)	1,065	1.04	2014
Tennessee Valley Authority	Browns Ferry 3	BWR (GE)	1,065	1.04	2016
Tennessee Valley Authority	Sequoya 1	PWR (West.)	1,122	1.08	2020
Tennessee Valley Authority	Sequoya 2	PWR (West.)	1,122	1.08	2021
Tennessee Valley Authority	Watts Bar 1	PWR (West.)	1,170	1.11	2030
Tennessee Valley Authority	Watts Bar 2	PWR (West.)	1,170	1.11	2032
Toledo Edison	Davis-Besse	PWR (B&W)	873	0.92	2037

Electric Utility Name	Reactor Name	Reactor Type	Power Rating (MWe)	Scaling Factor*	Projected Year of Shutdown
TU Electric	Comanche Peak 1	PWR (West.)	1,150	1.10	2030
TU Electric	Comanche Peak 2	PWR (West.)	1,150	1.10	2030
Union Electric	Callaway	PWR (West.)	1,125	1.08	2024
Vermont Yankee Nuclear	Vermont Yankee	BWR (GE)	496	0.63	2012
Virginia Power	North Anna 1	PWR (West.)	911	0.94	2018
Virginia Power	North Anna 2	PWR (West.)	909	0.94	2020
Virginia Power	Surry 1	PWR (West.)	781	0.85	2012
Virginia Power	Surry 2	PWR (West.)	781	0.85	2013
Washington Public Power	Washington Nuclear 2	BWR (GE)	1,100	1.07	2023
Washington Public Power	Washington Nuclear 3	PWR (CE)	1,250	1.16	2040
Wisconsin Electric	Point Beach 1	PWR (West.)	495	0.63	2010
Wisconsin Electric	Point Beach 2	PWR (West.)	495	0.63	2013
Wisconsin Public Service	Kewaunee	PWR (West.)	519	0.65	2014
Yankee Atomic Electric	Yankee Rowe	PWR (West.)	167	0.30	1992

ADDENDUM #2

RADIATION SURVEY DATA FOR HUMBOLDT BAY

Radiation Survey-refueling Building[a]

Location		Dose Rate[b] mr/h		Contamination Levels (μci/100cm^2) Contact[c]		Smearable	
		Gamma[d]	Beta	Alpha	Beta-Gamma	Alpha	Beta-Gamma[e]
+12 ft Elevation	floor	10	<1	f	3.6E-2	3.9E-6	1.1E-3
	wall			f	9.8E-3	2.2E-6	3.3E-4
Access Shaft -2 ft El	floor	7[g]	h	f	1.6E-2	7.1E-6	1.5E-3
	wall			f	2.1E-3	f	2.7E-5
-14 ft El	floor	28	0	f	4.2E-3	4.7E-6	2.3E-3
	wall			f	2.4E-3	2.3E-6	7.6E-4
-24 ft El	floor	1[g]	h	f	3.1E-3	1.4E-5	2.4E-3
	wall			f	1.0E-3	f	f
-34 ft El	floor	1[g]	h	f	2.1E-3	1.2E-5	3.0E-3
	wall			f	f	f	f
-44 ft El	floor	7[g]	1.5	f	8.3E-2	4.5E-6	1.3E-3
	wall			f	1.0E-2	f	2.7E-5
-54 ft El	floor	18	1.1	f	1.2E-1	4.5E-6	1.2E-3
	wall			f	2.1E-2	f	f
-66 ft El	floor	12	0	f	1.4E-1	2.3E-6	6.1E-4
	wall			f	6.4E-2	f	f
Cleanup HX Room -2 ft El	floor	65	0	f	1.0E-1	2.1E-5	9.4E-3
	wall			f	4.2E-2	f	1.9E-5
Cleanup Demin Room -2 ft El	floor	6	1.5	f	2.1E-1	1.0E-4	4.2E-2
	wall			f	2.1E-3	2.0E-6	3.5E-4
Shutdown HX Room -14 ft El	floor	55	1.1	f	f	3.7E-6	2.8E-3
	wall			f	2.1E-2	2.8E-7	2.0E-5
West Wing -66 ft El	floor	110	7.5	f	f	1.2E-5	2.7E-3
	wall			f	9.6E-2	5.6E-7	f
Under Reactor -66 ft El	floor	23	21	1.7E-3	2.0E+0	9.0E-4	3.3E-1
	wall			f	3.2E-2	6.5E-5	4.4E-3

Radiation Survey-refueling Building[a]

Location		Dose Rate[b]		Contamination Levels (μci/100cm^2)			
		mr/h		Contact[c]		Smearable	
		Gamma[d]	Beta	Alpha	Beta-Gamma	Alpha	Beta-Gamma[e]
New Fuel Vault +0 ft El	floor	5	47	3.4E-4	2.3E+0	1.9E-5	5.4E-3
	wall			f	f	1.1E-6	6.3E-4
TBDT Area -14 ft El	floor	23	35	f	1.6E-1	4.2E-6	9.6E-4
	wall			f	3.4E+0	1.1E-6	9.1E-3

[a] Average values of PG&E Survey conducted May 1984 unless otherwise specified.

[b] Ion Chamber.

[c] Minimum Sensitivity
Alpha: Approximately 1E-4 μCi/100cm^2
Beta: Approximately 5E-3 μCi/100cm^2 for Cutie Pie
Approximately 2E-6 μCi/100cm^2 for HP-210

[d] Based on ^{137}Cs.

[e] Based on ^{90}Sr (10%), ^{60}Co (45%) and ^{137}CS (45%).

[f] Not detected.

[g] Previous survey.

[h] Data not recorded.

Radiation Survey-power Building[a]

Location		Dose Rate[b]		Contamination Levels (μci/100cm^2)			
		mr/h		Contact[c]		Smearable	
		Gamma[d]	Beta	Alpha	Beta-Gamma	Alpha	Beta-Gamma[e]
Cond. Demin. Cubicle	floor	11	0	f	3.2E-2	8.5E-6	1.4E-3
	wall			f	3.2E-2	f	9.7E-5
Cond. Demin. Regen. Room	floor	14	1.5	2.6E-4	3.5E-2	1.1E-5	2.7E-3
	wall			1.0E-3	7.1E-2	1.1E-5	1.5E-3
Cond. Demin. Op. Area	floor	14[g]	h	f	3.5E-3	1.4E-6	1.5E-4
	wall			f	8.8E-3	f	6.1E-5
Cond. Pump Room	floor	13[g]	h	f	f	2.0E-6	5.0E-4
	wall			f	f	f	2.8E-5
Air Ejector Room	floor	55	56	f	5.6E+0	1.7E-6	7.8E-2
	wall			f	f	h	1.5E-3
Condenser Area	floor	19	<1	f	6.0E-3	5.7E-7	5.7E-4
	wall			f	f	h	h
Pipe Tunnel	floor	15	1.5	f	4.7E-3	1.1E-6	2.9E-4
	wall			f	f	1.4E-7	2.1E-5
Feed Pump Room	floor	<1[g]	h	f	5.2E-4	f	8.4E-5
	wall			h	h	h	h
Seal Oil Room	floor	0.005[g]	h	f	f	f	2.1E-5
	wall			h	h	h	h
Turbine Enc +27 ft El	floor	<1.[g]	h	f	3.1E-3	8.5E-7	1.2E-4
	wall			f	4.2E-3	2.8E-7	f
Turbine Washdown Area +27 ft El	floor	<1[g]	h	f	1.0E-3	1.7E-6	6.1E-5
Hot Lab	floor	<1[g]	h	f	1.2E-2	f	7.3E-5
Laundry/ Demin Area +27 ft El	floor	<1[g]	h	f	2.6E-3	4.3E-7	7.7E-5
Laundry/ Hot Lab +34 ft El	floor	h	h	f	1.0E-3	f	2.0E-4

Radiation Survey-power Building[a]

Notes:

[a] Average values of PG&E Survey conducted May 1984 unless otherwise specified.

[b] Ion Chamber.

[c] Minimum Sensitivity
Alpha: Approximately 1E-4 μCi/100cm^2
Beta: Approximately 5E-3 μCi/100cm^2 for Cutie Pie
Approximately 2E-3 μCi/100cm^2 for HP-210

[d] Based on ^{137}Cs.

[e] Based on ^{90}Sr (10%), ^{60}Co (45%) and ^{137}Cs (45%).

[f] Not detected.

[g] Previous survey.

[h] Data not recorded.

APPENDIX A

CHARACTERIZATION OF SCRAP METAL INVENTORIES AT U.S. NUCLEAR POWER PLANTS

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CHARACTERIZATION OF SCRAP METAL INVENTORIES AT U.S. NUCLEAR POWER PLANTS

1.0 INTRODUCTION

The U.S. commercial nuclear power industry is represented by 123 reactor plants. At present, eight reactors have been shutdown; and in the next two to three decades, most of the reactors currently in operations will have reached their projected forty-year lifetime.

With the publication of the NRC's Decommissioning Rule in June 1988 (US NRC 1988), owners and/or operators of licensed nuclear power plants are required to prepare and submit plans and cost estimates for decommissioning their facilities to the NRC for review. Decommissioning, as defined in the rule, means to remove nuclear facilities safely from service and to reduce radioactivity to a level that permits release of the property for unrestricted use and termination of the license. The decommissioning rule applies to the site, buildings, and contents and equipment. Currently, several utilities have submitted a decommissioning plan to the NRC for review.

Historically, the NRC has defined three classifications for decommissioning of nuclear facilities:

- DECON is defined by the NRC as "the alternative in which the equipment, structures, and portions of a facility and site containing radioactive contaminants are removed or decontaminated to a level that permits the property to be released for unrestricted use shortly after cessation of operations."

- SAFSTOR is defined as "the alternative in which the nuclear facility is placed and maintained in a condition that allows the nuclear facility to be safely stored and subsequently decontaminated (deferred dismantlement) to levels that permit release for unrestricted use."

  The SAFSTOR decommissioning alternative provides a condition that ensures public health and safety, from residual radioactivity remaining at the site, without the need for extensive modification to the facility. Systems not required to be operational for fuel storage, maintenance, and surveillance purposes during the dormancy period are to be drained, de-energized, and secured.

- ENTOMB is defined as "the alternative in which radioactive contaminants are encased in a structurally long-lived material, such as concrete; the entombed structure is appropriately maintained and continued surveillance is carried out until the radioactive material decays to a level permitting unrestricted release of the property."

Over the years, the basic concept of the three alternatives has remained unchanged. However, because of the accumulated inventory of spent nuclear fuel (SNF) in the reactor storage pool and the requirement for about seven years of pool storage for the SNF before transfer to dry storage, the timing and steps in the process for each alternative have had to be adjusted to reflect present conditions. For the DECON alternative, it is assumed that the owner has a strong incentive to decontaminate and dismantle the retired reactor facility as promptly as possible, thus necessitating transfer of the stored SNF from the pool to a dry storage facility on the reactor site. While continued storage of SNF in the pool is acceptable, the Part 50 license could not be terminated until the pool had been emptied, and only limited amounts of decontamination and dismantlement of the facility would be required. This option also assumes that an acceptable dry transfer system will be available to remove the SNF from the dry storage facility and to place it into licensed transport casks when the time comes for DOE to accept the SNF for disposal at a high level waste repository.

In addition, the amended regulation stipulates that alternatives, which significantly delay completion of decommissioning, such as use of a storage period, will be acceptable if sufficient benefit results. The Commission indicated that a storage period of up to 50 years and a total of 60 years between shutdown and decommissioning is a reasonable option for decommissioning a light water reactor. In selecting 60 years as an acceptable period of time for decommissioning of a nuclear power reactor, the Commission considered the amount of radioactive decay likely to occur during an approximate 50-year storage period and the time required to dismantle the facility.

In summary, the need to adequately cool the high burnup assemblies from the final fuel core in the pool for up to seven years and the regulatory requirements that critical support systems be maintained in operable conditions, the time between shutdown, decontamination, and the earliest date of dismantling efforts that would generate scrap metal is likely to be about 10 years. This interval may extend up to 60 years under the decommissioning alternative SAFSTOR. Through natural decay, a longer time interval has the obvious impact of greatly reducing radionuclide inventories. However, a simple inverse correlation between reduced levels of contamination and increased quantities of scrap metal suitable for recycling cannot be inferred. It is likely that for most scrap metal, the reduced contamination induced by natural decay may merely impact the choice of decontamination method and/or decontamination effort required to meet a future standard. For example, a storage period that reduces beta/gamma surface contamination of 1.0×10^7 dpm/100 cm^2 at 10 years post-shutdown to 1.0×10^5 dpm/100 cm^2 (i.e., a 100-fold reduction) would still require substantial decontamination in order to meet current standards defined by NRC Regulatory Guide 1.86. However, the reduced contamination level that is likely to be dominated by Cs-137 may affect the method and level of effort required for successful decontamination.

The potential use of scrap metal for recycling is, therefore, dictated by the cost-effectiveness with which materials can be decontaminated to levels deemed acceptable for unrestricted use (or for specified restricted use(s)). Estimates of scrap metal quantities must consider starting levels of contamination and whether the contamination is surficial or volumetrically distributed.

Residual radioactivity pertaining to reactor components/systems and building structures is generally grouped as (1) activation products that are distributed volumetrically, (2) activation and fission products in the form of corrosion films deposited on internal surfaces, and (3) contamination of external surfaces that result from the deposition of liquid and airborne contaminants associated with steam, primary coolant, and radioactive waste streams.

Most of the metal scrap available from the complete dismantling of a nuclear power plant is not expected to be radioactive. The non-radioactive scrap includes the large quantities of structural metals and support systems that have not been exposed to radioactivity during reactor operations. Conversely, some metal components will undoubtedly be contaminated so as to render them unsuitable for recycling.

2.0 CHARACTERISTICS OF REFERENCE REACTOR FACILITIES

A crucial factor affecting the quantity of metal and associated contamination levels is the basic design of the reactor. The two types of reactors used in the United States are the pressurized water reactor (PWR) and the boiling water reactor (BWR). Of the 123 U.S. reactor units, 40 are BWRs manufactured by General Electric (GE) and 83 are PWRs manufactured by Westinghouse (W), Combustion Engineering (CE), and Babcock and Wilcox (B&W). Addendum #1 provides a complete listing of U.S. reactors along with demographic data that includes projected year of shutdown.

In the 1976-1980 time frame, two studies were carried out for the U.S. Nuclear Regulatory Commission by the Pacific Northwest Laboratory (PNL) that examined the technology, safety, and costs of decommissioning large reference nuclear power reactor plants. Those studies, NUREG/CR-0130 and NUREG/CR-0672 for a reference PWR and reference BWR, respectively, reflected the industrial and regulatory situation of the time.

To support the final Decommissioning Rule issued in 1988, the earlier PNL studies have been updated with the recent issuance of NUREG/CR-5884, Revised Analyses of Decommissioning for the Reference Pressurized Water Reactor Station and NUREG/CR-6174, Revised Analyses of Decommissioning for the Reference Boiling Water Reactor Power Station. These four NUREG reports along with several other NRC reports and select decommissioning plans on file with the Commission represent the primary source of information used to characterize Reference PWR and BWR facilities and to derive estimates of scrap metal inventories for the industry at large.

2.1 Reference PWR Design and Building Structures

The facility described in this section is the 3,500 MWt (1,175 MWe) Trojan Nuclear Plant (TNP) at Rainier, Oregon - operated by the Portland General Electric Company (PGE). Designed by Westinghouse, this reactor is considered a typical pressurized water reactor that has been cited as Reference PWR (NUREG/CR-0130; NUREG/CR-5884).

The NRC granted the operating license for the TNP on November 21, 1975, and the plant formally began commercial operation on March 20, 1976. TNP's operating license was scheduled to expire on February 8, 2011. However, on November 9, 1992, the TNP was shutdown when a leak in the "B" steam generator was detected and the licensee notified the NRC of its decision to permanently cease operations in January 1993. Following the transfer of spent fuel from the reactor vessel to the spent fuel pool in May of 1993, TNP's Operating license was reduced to a Possession Only license. TNP's 17-year operating period encompassed 14 fuel cycles and approximately 3,300 effective full-power days. In the decommissioning plan submitted by PGE, the licensee has proposed the DECON approach with a five-year delay period prior to decontamination and dismantlement.

In a PWR, the circulating primary coolant water is heated by the nuclear fuel core but is prevented from boiling by a pressurizer, which maintains a pressure of about 2,000 psi. The principal systems and components of the nuclear steam supply system are illustrated in Figure A2-1. Components of interest are the reactor vessel, which contains the fuel and coolant and the reactor coolant system (RCS). The reactor vessel also contains internal support structures (not depicted) that constrain the fuel assemblies, direct coolant flow, guide in-core instrumentation, and provide some neutron shielding. The RCS consists of four loops for transferring heat from the reactor's primary coolant to the secondary coolant system. Each loop consists of a steam generator, a reactor coolant pump, and connecting piping. Steam generated from secondary feedwater is passed through the turbine, condensed back to water by the condenser, and recycled.

Also included in the primary loop is a small side stream of water that is directed to the chemical volume and control system (CVCS). This CVCS provides chemical and radioactive cleanup of the primary water through demineralizers and evaporators. The primary water is reduced in both pressure and temperature by the CVCS before being processed; therefore, the CVCS is often referred to as the letdown system. The water processed through the CVCS is returned to the primary water loops by the charging pumps. Note that the primary water processed through the CVCS is brought through the containment boundary or out of the containment building, but the primary water providing the heat transfer to the steam generators does not pass through the containment boundary.

As shown in Figure A2-1, high levels of internally contaminated components for a pressurized water reactor are those associated with the primary coolant water system. Low-level contamination of the secondary loop is a result of steam generator tube leakage in which limited quantities of primary coolant are introduced into the recirculating steam/water. Other major contaminated systems of PWRs not shown in Figure A2-1 include the radioactive waste handling system and the spent fuel storage system.

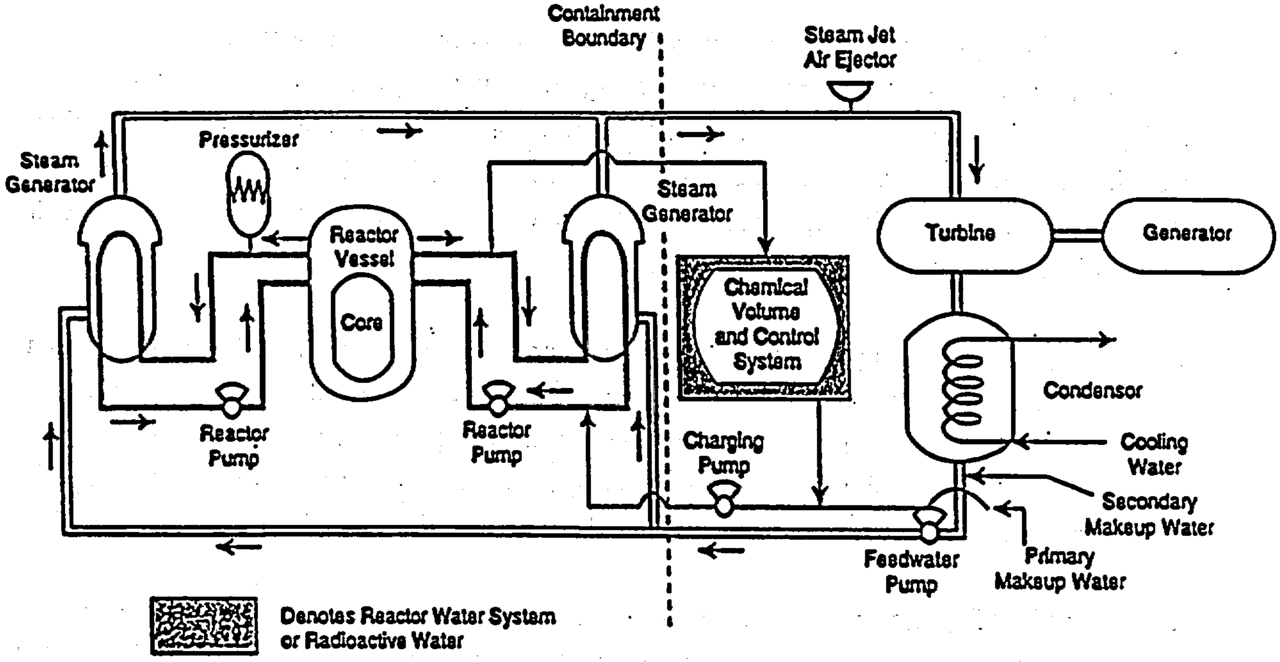

Figure A2-1. Pressurized Water Reactor

The principal structures requiring decontamination for license termination at the Reference PWR power station are the (1) Reactor Building, (2) Fuel Building, and (3) Auxiliary Building. In addition to housing major plant systems, all three buildings contain contaminated systems and substantial quantities of contaminated structural metals that are potentially available for recycling.

Reactor Building. The reactor building houses the nuclear steam supply system. Since its primary purpose is to provide a leak tight enclosure for normal as well as accident conditions, it is frequently referred to as the containment building. Major interior structures include the biological shield, pressurizer cubicles, and a steel-lined refueling cavity. Supports for equipment, operating decks, access stairways, grates, and platforms are also part of the containment structure internals.

The Reactor Building is in the shape of a right circular cylinder approximately 64 m tall and 22.5 m in diameter. It has a hemispherical dome, a flat base slab with a central cavity and instrumentation tunnel.

The Fuel Building - approximately 27 m tall and 19 by 54 m wide - is a steel and reinforced concrete structure with four floors. This building contains the spent-fuel storage pool and its cooling system, much of the Chemical and Volume Control System, and the solid radioactive waste handling equipment. Major steel structural components include fuel storage racks and liner, supports structures for fuel handling, and components, ducts, and piping associated with air conditioning, heating, cooling, and ventilation.

The Auxiliary Building - approximately 30 m tall and lateral dimensions of 19 by 35 m - is a steel and reinforced concrete structure with two floors below grade and four floors above grade. Principal systems contained in the Auxiliary Building include the liquid radioactive waste treatment systems, the filter and ion exchanger vaults, waste gas treatment system, and the ventilation equipment for the Containment, Fuel, and Auxiliary Buildings.

Other major building structures with substantial inventories of metals include the Control Building and Turbine Building. The principal contents of the Control Building are the reactor control room, process and personnel facilities. The principal systems contained in the Turbine Building are the turbine generator, condensers, associated power production equipment, steam generator auxiliary pumps, and emergency diesel generator units.

Barring major system failure(s) (e.g., steam generator failure) most scrap metal derived from these systems can be assumed to be free of contamination and can, therefore, be excluded from estimates of scrap metal inventories.

2.2 Reference BWR Reactor Design and Building Structures

The 3,320 MWt (1,155 MWe) Washington Public Power Supply System (WPPSS) Nuclear Project No. 2 located near Richland, Washington has historically been cited as the Reference BWR facility (NUREG/CR-0672; NUREG/CR-6174).

The design of a BWR (see Figure A2-2) is simpler than a PWR inasmuch as the reactor coolant water is maintained near atmospheric pressure and boiled to generate steam. This allows the coolant water to directly drive the turbine. Thereafter, the steam is cooled in the condenser and returned to the reactor vessel to repeat the cycle. In a BWR, the contaminated reactor coolant comes in contact with most major BWR components, including the reactor vessel and water piping, steam turbine, steam condenser, feedwater system, reactor water cleanup system, and steam jet air ejector system. As with the PWR, other major contaminated reactor systems include the radioactive waste treatment system and spent fuel storage system.

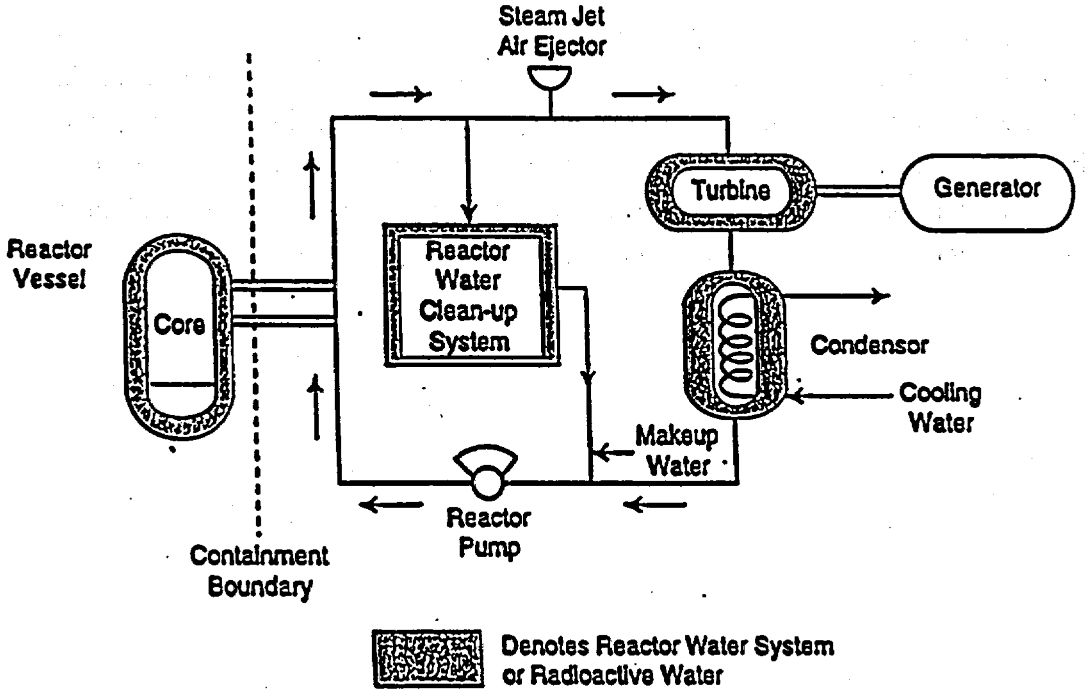

Figure A2-2. Boiling Water Reactor

The principal buildings requiring decontamination and dismantlement in order to obtain license termination at the reference BWR power station are the Reactor Building, the Turbine Generator Building, and the Radwaste and Control Building. These three buildings contain essentially all of the activated or radioactively contaminated material and equipment within the plant.

The Reactor Building contains the nuclear steam supply system and its supporting systems. It is constructed of reinforced concrete capped by metal siding and roofing supported by structural steel. The building surrounds the primary containment vessel, which is a free-standing steel pressure vessel. The exterior dimensions of the Reactor Building are approximately 42 m by 53 m in plan, 70 m above grade, and 10.6 m below grade to the bottom of the foundation.

The Turbine Building, which contains the power conversion system equipment and supporting systems, is constructed of reinformed concrete capped by steel-supported metal siding and roofing. This structure is approximately 60 m by 90 m in plan and 42.5 m high.

The Radwaste and Control Building houses, among other systems: the condenser off-gas treatment system, the radioactive liquid and solid waste systems, the condensate demineralizer system, the reactor water cleanup demineralizer system, and the fuel-pool cooling and cleanup demineralizer system. The building is constructed of reinforced concrete, structural steel, and metal siding and roofing. This structure is approximately 64 by 49 m in plan, 32 m in overall height, and stands as two full floors and one partial floor above the ground floor.

3.0 INVENTORIES OF RESIDUAL RADIOACTIVITY IN REFERENCE REACTOR FACILITIES

Significant quantities of radioactivity remain in a nuclear power station following reactor shutdown even when all spent nuclear fuel has been removed. Neutron-activated structural materials in and around the reactor pressure vessel contain the majority of residual radioactivity in relatively immobile condition. Lesser quantities involve radioactive corrosion products and fission products from failed fuel, which are transported throughout the station by the reactor coolant streams. The origin and mobility of radioactive nuclides following reactor shutdown leads to grouping of residual radioactivity into five categories of different binding matrices. These categories include:

1. Activated Stainless Steel - Reactor internals, composed of Type 304 stainless steel, become activated by neutrons from the core. Radionuclides have very high specific activities and are immobilized inside the corrosion-resistant metal.

2. Activated Carbon Steel - Reactor pressure vessels are made of SA533 carbon steel that becomes activated by neutrons bombardment. The specific activities are considerably lower than in the stainless steel internals; and the binding matrix is much less corrosion resistant.

3. Activated Structural Steel, Steel Rebar, and Concrete. In the reactor cavity, these components become activated from neutrons escaping from the reactor vessel. Significant activation occurs along approximately 15 feet of the reactor cavity vertically centered on the reactor core and to a depth of about 16 inches in the concrete.

4. Contaminated Internal Surfaces of Piping and Equipment - Activated corrosion and fission products from the fuel travel through the reactor coolant water system throughout the radioactive liquid systems in the plant. A portion forms a hard metallic oxide scale on the inside surfaces of pipes and equipment.

5. Contaminated External Surfaces - Surfaces may become contaminated over the lifetime of the plant primarily from leaks, spills and airborne migration of radionuclides

contained in the reactor coolant water (RCW). The specific activity of RCW is low, but the contamination is easily mobilized and may be widespread.

All of the neutron activated metals/materials are contained in the reactor pressure vessel, vessel internals, and structural components inside and within the concrete biological shield.

Total quantities and relative radionuclide composition of residual radioactivity are not only affected by reactor design (i.e., BWR versus PWR) but are also strongly influenced by numerous other factors inclusive of (1) fuel integrity, (2) rated generating capacity and total years of operation, (3) composition of metal alloys of reactor components and reactor coolant system, (4) coolant chemistry and water control measures, and (5) the performance of or failures of critical systems and their maintenance over the 40 years of operation.

Table A3-1 provides summary estimates of typical residual activities for each of the five major source categories. Inspection of data reveals the volumetrically activated stainless steel represents the overwhelming percentage of residual radioactivity. Much lower quantities are represented by volumetrically activated carbon steel and internal and external surface contamination consisting of activation and fission products. A more detailed discussion of residual radioactivity by source category is given below.

Table A3-1. Source of Residual Radioactivity and Curie Quantities at Reference BWR and PWR

Source Category	Quantities (Ci)	
	BWR[a]	PWR[b]
Activated Stainless Steel	6.6×10^6	4.8×10^6
Activated Carbon Steel	2.9×10^3	2.4×10^3
Activated Structural Components, Rebar, Metal Plates, I-Beams	1.2×10^3	1.2×10^3
Internal Surface Contamination of Piping and Equipment	8.5×10^3	4.8×10^3
External Contamination of Equipment, Floors, Walls, and Other Surfaces	1.1×10^2	(1.1×10^2)[c]

[a] NUREG/CR-0672
[b] NUREG/CR-0130
[c] Implied Value - NUREG-1496

3.1 Neutron Activated Reactor Components and Structural Materials

Contamination of reactor components and structural materials by neutron activation is the result of normal reactor operation. The interaction of neutrons with constituents of stainless steel, carbon steel, and concrete in and around the reactor vessel results in large quantities of in-situ radioactivity. Major radionuclides include Cr-51, Mn-54, Fe-55, Fe-59, Co-58, Ni-59, and Ni-63. The activity concentration or buildup of a particular radionuclide among materials subject to neutron activation is highly variable and depends upon (1) the concentration of the parent isotope and its neutron cross-section, (2) the physical half-life of the radioactive specie(s), (3) the neutron flux level at that location, and (4) the duration of neutron exposure.

Reference BWR. The average radioactivity concentrations and estimated quantities of radioactivity for Reference BWR structural components with significant amounts of neutron activation are listed in Tables A3-2.

Table A3-2. Estimated Radioactivity of Neutron-Activated Reactor Components in a BWR

Component (Quantity)	Volume-Averaged Radioactive Concentration (Ci/m^3)	Estimated Total Radioactivity (Ci)
Core Shroud (1)	1.68×10^6	6.30×10^6
Jet Pump Assembly (10)	2.62×10^4	2.00×10^3
Reactor Vessel (1)		2.16×10^3
Cladding	1.07×10^3	
Shell Wall	1.12×10^2	
Steam Separator Assembly (1)		9.60×10^3
Shroud Head Plant	1.03×10^4	
Steam Separator Risers	2.53×10^3	
Top Fuel Guide (1)	9.71×10^4	3.01×10^4
Orificed Fuel Support (193)	1.01×10^3	7.01×10^2
Core Support Plate (1)	2.56×10^2	6.50×10^2
Incore Instrument Strings (55)	7.67×10^5	1.10×10^4
Control Rod (185)	5.11×10^5	1.78×10^5
Control Rod Guide Tube (185)	2.16×10^2	9.47×10^1
TOTAL		6.55×10^6

Source: NUREG/CR-0672

The Reference BWR reactor vessel is fabricated of SA533 carbon steel about 171-mm thick and is clad internally with 3 mm of Type 304 stainless steel. The total mass of the empty vessel is about 750 metric tons. The major internal components include the fuel core support structure; steam separators and dryers; coolant recirculation jet pumps; control rod guide tubes; distribution piping for feedwater, core sprays, and liquid control; in-core instrumentation and miscellaneous other components. Collectively, these internals, made of stainless steel, represent about 250 metric tons.

Reference PWR. The right circular cylinder of the Reference PWR is constructed of carbon steel, about 216 mm in thickness and is clad on the inside with stainless steel or Inconel having a thickness of about 4 mm. The approximate dimensions of the vessel are 12.6 m high, 4.6 m outer diameter. The vessel weighs about 400 metric tons.

The vessel internal structures support and constrain the fuel assemblies, direct coolant flow, guide in-core instrumentation, and provide some neutron shielding. The principal components are: the lower core support assembly, which includes the core barrel and shroud, with neutron shield pads and the lower core plate and supporting structure; and the upper core support and in-core instrumentation support assemblies. These structures are made of 304 stainless steel and have a total weight of about 190 metric tons.

Based on 40 years of facility operation that assumes 30 effective full-power years (EFPY) of reactor operation, the total number of curies contained in the activated vessel and internals is estimated at 4.8 million curies (Table A3-3). Extra-vessel materials subject to significant neutron activation (~10 Ci) includes the reactor cavity steel liner and a limited quantity of reinforcement steel (rebar). Additionally, about 1,200 Ci of radioactivity are estimated for the concrete bioshield.

Table A3-3. Estimated Radioactivity of Neutron-Activated Reactor Components in a PWR

Component	Volume-Averaged Radioactive Concentration (Ci/m^3)	Radioactivity per Component (Ci)
Shroud	2.97×10^6	3.43×10^6
Lower 4.7 m of Core Barrel	3.07×10^5	6.52×10^5
Thermal Shield	1.45×10^5	1.46×10^5
Vessel Inner Cladding	7.73×10^3	1.50×10^3
Lower 5.02 m of Vessel Wall	9.04×10^2	1.76×10^4
Upper Grid Plate	4.20×10^4	2.43×10^4
Lower Grid Plate	1.12×10^6	5.53×10^5
TOTAL		4.82×10^6

Source: NUREG/CR-0130

The projected estimates of Table A3-4 in behalf of the Reference PWR (i.e., Trojan Nuclear Plant) made in 1978 can be compared to the more current estimates contained in that plant's Decommissioning Plan (submitted to the NRC in 1996). Table A3-4 identifies revised calculated inventories of activation products for 1993 or one year after shutdown. The recalculated value of about 4.2 million curies is about 13% lower than the original estimate of 4.8 million curies and principally reflects the difference between 17 years of actual plant operation and the initial projection of 40 years.

Table A3-4. Activation Levels at Trojan Nuclear Plant (one year after shutdown)

Location	Activity	
	Terabecquerels	Curies
Reactor Vessel	230	6,200
Reactor Vessel Internals	154,000	4,160,000
Vessel Clad and Insulation	880	23,700
Bioshield Wall	31	830
Total	155,000	4,190,000

The considerably higher calculated radioactivity values for a Reference BWR primarily reflect the larger size and weight of the vessel and its internals.

For both PWR and BWR plants, the range of activity concentrations among individual reactor components at time of shutdown is likely to vary over several orders of magnitude. Nevertheless, even those components with the lowest activity concentrations would still exceed residual activity levels far in excess of any conceivable levels that would permit recycling for unrestricted use. (Note: At a density of 8,000 kg/m^3, a cubic meter of steel containing 1 curie represents a specific activity concentration of 275,000 pCi per gram.) Furthermore, these components also exhibit high levels of interior surface contamination. While surface contamination is potentially removable, the volumetrically-distributed activation products are not.

For this reason, the reactor vessel and all internal components identified in Tables A3-2 and A3-3 must be excluded from plant material inventories with regard to recycling. Excluded for similar reasons are select metal components used for structural support and reinforcement (i.e., rebar, I-beams, and floor and reactor cavity liner plates) that exhibit significant levels of activation products.

Scrap metal that has the potential for recycling is, therefore, limited to reactor systems and structural components where contamination is limited to interior and exterior surfaces.

3.2 Internal Surface Contamination of Equipment and Piping

Activated corrosion products from structural materials in contact with the reactor coolant and fission products from leaking fuel contribute to the presence of radioactivity in reactor coolant streams during plant operation. Although most of these radionuclides are removed through filtration and demineralization by a plant's chemical and volume control systems (letdown cleanup system), a smaller component escapes removal. With time, some of the radionuclides, principally the neutron-activated insoluble corrosion products, tend to deposit on inner surfaces of equipment and piping systems. The metal oxide layer consists primarily of iron, chromium, and nickel with smaller, but radiologically significant, quantities of cobalt, manganese, and zinc. This section characterizes the mixture of internal surface contaminants and their relative distribution within major components associated with BWR and PWR power plants.

3.2.1 Measurements of Internal Surface Contaminants at Six Nuclear Power Plants

In a 1986 NRC study, three PWRs and three BWRs were assessed with regard to residual inventories and distributions of long-lived radionuclides following plant shutdown (NUREG/CR-4289). Residual concentrations in the various plant systems decreased in the following order: (1) primary coolant loop, (2) radwaste handling system, and (3) secondary coolant loop in PWRs and condensate system in BWRs. Table A3-5 identifies total estimated contamination inventories for the six reactor facilities examined, as well as the electrical ratings, and the approximate number of operational years for the units at the time of inventory assessments. The operational periods ranged from 8.3 years for Turkey Point Unit 3 to slightly over 18 years for Dresden Unit One.

Table A3-5. Comparison of Residual Radionuclide Inventories and Operating Parameters for the Six Nuclear Generating Stations Examined*

Stations	Total Inventory (Curies)	Years of Operation	MWe	Reactor Type
Humboldt Bay	600	13	63	BWR
Dresden-1	2,350	18.3	210	BWR
Monticello	514	10	550	BWR
Indian Point-1	1,050	11	170	PWR
Turkey Point-3	2,580	8.3	660	PWR
Rancho Seco	4,470	8.8	935	PWR

Source: NUREG/CR-4289

* Inventory includes radionuclides with half-lives greater than 245 days (Zn-65); inventory estimates do not include the highly activated metal components of the reactor pressure vessel and internals and activated concrete.

The relative radionuclide composition of internally contaminated surfaces observed at the reactor sites also showed considerable variation (Table A3-6). Fluctuations in compositions were influenced by numerous factors including: (1) the elapsed time since reactor shutdown; (2) rated generating capacity; (3) materials of construction for the operating systems; (4) reactor type, e.g., PWR versus BWR; (5) coolant chemistry and corrosion control; (6) fuel integrity during operations; and (7) episodic equipment failure and leakage of contaminated liquids.

Table A3-6. Radionuclide Composition of Internal Surface Contamination*

Radionuclide	Composition in Percent of Total Activity Decay Corrected to Shutdown Date					
	BWRs			PWRs		
	Humboldt Bay	Dresden-1	Monticello	Indian Point-1	Turkey Point-3	Rancho Seco
Mn-54	3	0.9	1	4	0.4	4
Fe-55	90	28	1	67	31	28
Co-57	—	—	—	—	43	24
Co-60	6	46	11	15	24	18
Ni-59	—	0.09	—	0.02	0.004	0.1
Ni-63	0.2	5	0.04	2	0.1	19
Zn-65	—	19	84	11	1	0.09
Sr-90	0.004	0.007	0.002	0.0007	0.0008	< 0.01
Nb-94	< 0.004	< 0.003	< 0.1	0.0008	< 0.004	< 0.004
Tc-99	3×10^{-4}	4×10^{-5}	8×10^{-5}	8×10^{-5}	0.008	< 0.005
Ag-110m	—	—	—	—	—	4
I-129	$< 3 \times 10^{-6}$	$< 1 \times 10^{-5}$	$< 1 \times 10^{-6}$	2×10^{-5}	< 0.003	$< 1 \times 10^{-5}$
Cs-137	0.5	0.04	2	0.5	—	0.4
Ce-144	—	1	—	—	0.2	< 0.04
TRU**	0.005	0.1	0.008	0.002	0.006	0.001
Total Plant Inventory (Curies)	596	2,350	448	1,070	2,580	4,460

Source: NUREG/CR-4289

* Excludes highly activated metal components of the reactor pressure vessel and internals, and activated concrete.

** Transuranic alpha-emitting radionuclides with half-lives greater than 5 years, include Pu-238, Pu-239, Pu-240, Am-241, Am-243, and Cm-244.

Inventories only include the radioactive contamination of corrosion film and crud on surfaces of the various plant systems and do not include the highly activated components of the pressure vessel. The most abundant radionuclides included Mn-54, Fe-55, Co-58, Co-60, and Ni-63. Zinc-65 was present in relatively high concentrations in BWR corrosion film samples. However, Fe-55 and Co-57/Co-60 were the most abundant radionuclides at all stations except Monticello. These two radionuclides constituted over 95% of the estimated inventories at Humboldt Bay and Turkey Point. At Indian Point Unit One, Dresden Unit One, Turkey Point Unit Three, and Rancho Seco, they accounted for 82, 74, 98, and 70%, respectively, of the total estimated inventory. Although Fe-55 and Co-60 accounted for the majority of the inventory (greater than 60% at five of the six stations), the relationship between the two radionuclides was quite variable.

The transuranic radionuclides (Pu-238, Pu-239, Pu-240, Am-241, Cm-242, and Cm-244) constituted percentages of the total inventory ranging from 0.001% at Rancho Seco to 0.1% at Dresden Unit One.

Secondary coolant loops in PWRs and condensate systems in BWRs contained much lower radionuclide concentrations than observed in primary loop or feedwater samples. Typically, radionuclide concentrations were two or more orders of magnitude lower in secondary system samples.

As expected, the steam generators contained the single largest repository of internally deposited radionuclides at the PWR stations examined (Table A3-7). The percentages of the total residual radionuclide inventories in the steam generators were 77, 89, and 94% for Indian Point One, Turkey Point Unit 3, and Rancho Seco, respectively. The other repository of significance in a PWR is the radwaste system, which typically contained 5 to 10% of the total residual inventory.

Table A3-7. Distribution in Percent of the Radionuclide Inventory Estimates for Three Pressurized Water Reactors

	Turkey Point-2	Indian Point-1	Rancho Seco	PWR Average
Steam Generators	89	77	94	86.7
Pressurizer	0.5	0.5	0.33	0.4
RCS Piping	0.9	2.6	0.71	1.4
Piping (Except RCS)	< 0.01	14	< 0.01	4.7
Secondary Systems	0.1	0.2	0.05	0.1
Radwaste	9.2	7	5	7.1

Source: NUREG/CR-4289

3.2.2 Internal Surface Contamination Levels Reported in Decommissioning Plans

Due to premature shutdown or projected shutdown within the next few years, only a small number of commercial reactor facilities have submitted a Decommissioning Plan to the NRC for review. Summarized below are system-specific internal contamination levels reported for one BWR and two PWRs.

Big Rock Point Nuclear Plant. The Big Rock Point Nuclear Plant is a small (67 MWe) BWR designed by the General Electric Company and constructed by Bechtel Power Corporation. Owned and operated by Consumers Power Company, the plant started commercial operation in March of 1963 and is projected for shutdown in the year 2000. Table A3-8 presents summary data of systems internally contaminated.

Table A3-8. Internal Contamination Levels of Big Point Nuclear Plant at Shutdown

System	Contamination (dpm/100 cm^2)	System	Contamination (dpm/100 cm^2)
Liquid Rad Waste Tanks	3.0E+10	Resin Transfer System	3.0E+07
Nuclear Steam Supply	9.0E+09	Off-gas System	3.0E+07
RDS	3.0E+09	Control Rod Drive	6.0E+06
Main Steam System	4.0E+08	Rad Waste Storage	9.0E+05
Fuel Pool	4.0E+08	Fuel Handling Equip	7.0E+05
Liquid Radwaste System	4.0E+08	Heating & Cooling Sys	3.0E+05
Condensate System	5.0E+07		

San Onofre Nuclear Generation Station Unit 1 (SONGS 1). SONGS 1 is a 436-mwe PWR reactor that started operation in 1968. As a result of an agreement with the California Public Utility Commission, operation of SONGS 1 was permanently discontinued on November 30, 1992 at the end of Fuel Cycle No. 11. A preliminary Decommissioning Plan, submitted to the NRC on December 1, 1992, proposed to maintain SONGS 1 in safe storage until the permanent shutdown of SONGS 2 and 3. SONGS 2 and 3 are licensed to operate until the year 2013.

In support of the SONGS 1 Decommissioning Plan, scoping surveys and analyses were performed that supplemented an existing radiological data base. The containment building, fuel storage building, and the radwaste/auxiliary building were identified as the principal structures containing significant levels of radioactivity within plant systems. Systems were grouped by contamination levels defined as (1) highly contaminated, (2) medium-level contaminated, and (3) low-level contaminated. Based on total radionuclide inventories and surface areas, an average contamination level for each of the three groupings was derived (Table A3-9).

Table A3-9. Plant Systems Radioactivity Levels at SONGS 1

Plant Systems	Estimated Total Surface Area	Estimated Level of Radioactivity (dpm/100 cm^2)	Estimated System Total Activity
<u>Highly Contaminated Systems</u>: LDS Letdown PAS Post Accident Sampling System PZR Pressurizer Relief RCS Reactor Coolant RHR Residual Heat Removal RSS Reactor Sampling SFP Spent Fuel Pool Cooling VCC Volume Control			
Total for Highly Contaminated Systems	1.26E+08 cm^2	3.6E+09	2.08E+03 Ci
<u>Medium Level Contaminated Systems</u>: BAS Boric Acid CWL Containment Water Level RCP RCP Seal Water RLC Radwaste Collection RMS Radiation Monitoring RWG Radwaste Gas RWL Radwaste Liquid CRS (Containment Spray) Recirc SIS Safety Injection			
Total for Medium Level Contaminated Systems	1.25E+08 cm^2	1.9E+06	1.08E+01 Ci

Table A3-9. Plant Systems Radioactivity Levels at SONGS 1 (Continued)

Plant Systems	Estimated Total Surface Area	Estimated Level of Radioactivity (dpm/100 cm^2)	Estimated System Total Activity
Low Level Contaminated Systems: AFW Auxiliary Feedwater CCW Component Cooling CND Condensate SHA Sphere Hydrazine Addition CSS Condensate Sampling CVD Condensate Vents & Drains CVI Cryogenics CWS Circulating Water FES Flash Evaporator FPS Fire Protection FSS Feed Sampling FWH Feed Water Heaters FWS Feedwater MSS Main Steam MVS Miscellaneous Ventilation PSC Turbine Sample Cooling SDW Service Water SWC Salt Water Cooling TCW Turbine Cooling			
Total for Low Level Contaminated Systems	3.18E+08 cm^2	8.3E+03	1.21E-02 Ci

Yankee Rowe. Yankee Rowe is a 167-mwe PWR with a startup date of August 19, 1960. It started commercial operation in July of 1961 and was shutdown in October of 1991 following 21 fuel cycles and 8,052 EFPD. In the 1993 decommissioning plan submitted to the NRC, systems with significant internal surface contamination were identified as shown in Table A3-10.

In reviewing the data of facilities that have submitted decommissioning plans, only limited conclusions can be drawn to derive Reference values due to issues that relate to (1) their limited years of operation, (2) abnormal events and operating conditions that prompted premature shutdown, and/or (3) size and design of the facilities.

Table A3-10. System Average Internal Contamination Levels for Yankee Rowe

System	Contamination (dpm/100 cm^2)	System	Contamination (dpm/100 cm^2)
Main Coolant	7.1E+09	Primary Plant Sampling	1.4E+06
Spent Fuel Cooling	3.3E+08	Safety Injection	1.4E+05
Waste Disposal	1.2E+07	Safe Shutdown	1.4E+05
Primary Plant Vent & Drain	1.2E+07	Vol. Control Heating & Cooling	1.2E+04
Charging & Volume Control	1.2E+07	Vol. Control Vent. & Purge	1.2E+04
Shutdown Cooling	1.2E+07	Post Accident H_2 Control	1.2E+04
Fuel Handling	1.7E+06	Chemical Shutdown	1.1E+04
Letdown/Purification	1.4E+06		

3.2.3 Levels of Internal Surface Contamination Derived for Reference BWR

Internal surface contamination levels in BWR systems and piping reflect the radionuclide concentrations in the reactor water, steam, and condensate. Summary inventory estimates of activity in corrosion film deposited on internal surfaces of equipment and piping are cited in NUREG/CR-0672 for a Reference BWR.

The radionuclide composition of corrosion films is shown in Table A3-11. About 86% of the estimated inventory at shutdown was due to two nuclides, Co-60 and Mn-54 (Co-60 contributed nearly half or 47% to the total inventory). It should be noted that internal surface deposited nuclides generally do not include large amounts of fission products. Although fission products do exist in the primary coolant, they are generally soluble and remain in solution rather than plate out along with neutron activated corrosion products. The buildup of coolant contaminants is controlled by the letdown system, which continuously removes both insoluble (particulate) and soluble contaminants.

Table A3-11. Relative Radionuclide Composition of Activated Corrosion Products of Reference BWR at Shutdown

Radionuclide	Fractional Radioactivity at Decay Times of:			
	Shutdown	10 Years	30 Years	50 Years
Cr-51	2.1×10^{-2}	—	—	—
Mn-54	3.9×10^{-1}	1.3×10^{-4}	—	—
Fe-59	2.5×10^{-2}	—	—	—
Co-58	9.3×10^{-3}	—	—	—
Co-60	4.7×10^{-1}	1.3×10^{-1}	9.1×10^{-3}	6.6×10^{-4}
Zn-65	6.1×10^{-3}	1.5×10^{-7}	—	—
Zr-95	4.0×10^{-3}	—	—	—
Nb-95	4.0×10^{-3}	—	—	—
Ru-103	2.3×10^{-3}	—	—	—
Ru-106	2.8×10^{-3}	2.7×10^{-6}	—	—
Cs-134	1.9×10^{-2}	—	—	—
Cs-137	3.4×10^{-2}	2.7×10^{-2}	1.7×10^{-2}	1.1×10^{-2}
Ce-141	3.0×10^{-3}	—	—	—
Ce-144	8.1×10^{-3}	8.8×10^{-7}	—	—
Totals	1.0	1.6×10^{-1}	2.6×10^{-2}	1.2×10^{-2}

The total radionuclide inventory has been estimated at 8,500 curies with 6,300 curies associated with internal equipment surfaces and the remaining 2,200 curies associated with internal piping surfaces (Table A3-12).

Table A3-12: Distribution of Activated Corrosion Products on Internal Surfaces in Reference BWR

Location	Surface Area (m^2)	Activity Level (Ci/m^2)	Total Deposited Activity (Ci)
Piping	3.2×10^{4}	6.8×10^{-2}	2.2×10^{3}
Equipment:			
Reactor Building	8.6×10^{3}	2.2×10^{-1}	1.9×10^{3}
Turbine Building	2.0×10^{5}	6.0×10^{-3}	1.2×10^{3}
Radwaste & Control	1.4×10^{3}	2.3×10^{0}	3.2×10^{3}
Total	2.4×10^{5}	2.6×10^{0}	8.5×10^{3}

For the residual equipment inventory of 6,300 curies, an estimated 30% was associated with equipment in the reactor building; about 19% was associated with the condenser and feed-water heaters located in the turbine building; and about 51% involved internal deposition of equipment in the radwaste and control building.

Of the 2,200 curies present in piping, approximately 56% was estimated to be associated with the reactor water piping and 44% with condensate piping. Presented below is a more thorough analysis of piping data.

<u>Contaminated Piping</u>. Internal surface contamination levels of BWR piping can be most useful when grouped according to direct or indirect contact with reactor water, steam/air, and condensate. Deposition levels for reactor water and condensate were based on empirical dose rate measurements that were correlated to contamination levels for a specific pipe size and schedule. A summary of measured dose rate data and derived deposition levels is shown in Table A3-13.

Table A3-13. Contact Dose Rate and Internal Radioactivity Deposition of BWR Piping

Piping Contact Media	Nominal Outside Diameter (mm)	Wall Thickness (mm)	Measured Contact Dose Rate (mR/hr)	Estimated Deposition Level (Ci/m^2)
Reactor Water	610	59.5	700	1.1
Steam/Air	914	20.4	70	0.005
Condensate	610	26.0	50	0.05

Table A3-14 provides a detailed accounting of radionuclide inventories derived for various size piping made of aluminum, carbon steel, and stainless steel in contact with reactor water, steam/air, or condensate.

Table A3-14. Estimates of Internal Contamination for Reference BWR Piping

Pipe Material (contact medium)	60 mm OD			152 mm OD			356 mm OD			533 mm OD			660 mm OD			914 mm OD		
	L (m)	A (m^2)	Act. (Ci)	L (m)	A (m^2)	Act. (Ci)	L (m)	A (m^2)	Act. (Ci)	L (m)	A (m^2)	Act. (Ci)	L (m)	A (m^2)	Act. (Ci)	L (m)	A (m^2)	Act. (Ci)
<u>Aluminum</u>																		
(Rx water)	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
(Steam/Air)	4,300	81	0.4	1,400	640	3.2	130	140	0.7	—	—	—	—	—	—	—	—	—
(Condensate)	—	—	—	14	6.7	0.3	—	—	—	—	—	—	—	—	—	—	—	—
<u>Carbon Steel</u>																		
(Rx water)	380	71	78	1,500	700	770	61	68	75	55	92	100	—	—	—	—	—	—
(Steam/Air)	1,200	220	1.1	1,800	880	4.4	5,600	6,300	32	1,200	2,000	10	950	200	9.8	440	1,300	6.3
(Condensate)	7,400	1,400	7.0	8,300	3,900	200	5,100	5,700	280	2,800	4,600	230	370	770	38	210	610	31
<u>Stainless Steel</u>																		
(Rx water)	8	1.5	1.6	34	16	18	61	68	75	55	92	100	—	—	—	—	—	—
(Steam/Air)	280	53	0.3	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
(Condensate)	7,000	1,300	66	1,600	780	39	220	240	12	—	—	—	—	—	—	—	—	—
Sub-totals	NA	3,126	154	NA	6,923	1,035	NA	12,516	475	NA	6,784	440	NA	9,740	48	NA	1,910	37

Total Activity: ~2,200 Ci
Total Area: 32,200 m^2

Average Contamination Level: 6.8 x 10^{-2} Ci/m^2
(1.5 x 10^9 dpm/100 cm^2)

Contaminated Equipment. Contamination on internal surfaces of BWR equipment in contact with reactor water was based on measurements taken for the heat exchanger in the reactor water cleanup system. In general, equipment in contact with steam or condensate was assumed to reach the same levels as previously cited for BWR piping. Exceptions were the lower values assigned to steam surfaces for the turbine and feedwater heaters. Table A3-15 provides estimates of radioactivity deposition levels assigned to BWR equipment.

Table A3-16 identifies the major system components and radionuclides inventories based on location and contact with reactor water, steam, condensate, and radwaste.

Table A3-15. Summary of Internal Radioactivity Levels in BWR Equipment

Equipment Category	Radioactivity Deposition Level[1] (Ci/m^2)
Reactor Water Equipment	3.6×10^{-1}
Steam Equipment	5.0×10^{-3}
Turbine	5.0×10^{-4}
Condensate Equipment	5.0×10^{-2}
Main Condenser	5.0×10^{-3}
Feedwater Heaters	5.0×10^{-3}
Concentrated Waste Tanks/Equipment	5.0×10^{-0}

[1] Note: 1 Ci/m^2 = 2.2×10^{10} dpm/100 cm^2

Table A3-16. Estimated Internal Radioactivity in BWR Systems

Building/System	Total Estimated Internal Surface Area (m^2)	Radioactivity Deposition Level[1] (Ci/m^2)	Deposited Radioactivity (Ci)
Reactor Building			
Fuel Pool Exchangers	8.0×10^{2}	5.0×10^{-2}	4.0×10^{1}
Skimmer Surge Tanks	1.0×10^{2}	5.0×10^{-2}	5.0×10^{1}
Fuel Pool, Rx Wall, Dryer & Sep. Pool	1.4×10^{3}	5.0×10^{-2}	7.0×10^{1}
RBCC Water Heat Exchangers	1.8×10^{3}	5.0×10^{-2}	9.0×10^{1}
RMCU Regenerative Heat Exchangers	2.5×10^{2}	3.6×10^{-1}	9.0×10^{1}
RWCU Nonregenerative Heat Exchangers	1.7×10^{2}	3.6×10^{-1}	6.0×10^{1}
RHR Heat Exchangers	1.5×10^{3}	3.6×10^{-1}	5.4×10^{2}
Reactor Vessel	2.6×10^{3}	3.6×10^{-1}	9.4×10^{2}
Total	8.6×10^{3}		1.9×10^{3}

Table A3-16. Estimated Internal Radioactivity in BWR Systems (Continued)

Building/System	Total Estimated Internal Surface Area (m^2)	Radioactivity Deposition Level[1] (Ci/m^2)	Deposited Radioactivity (Ci)
Turbine Generator Building			
Main Condenser	7.9×10^4	5.0×10^{-3}	3.9×10^2
Steam Jet Air Ejector Condenser	1.6×10^3	5.0×10^{-2}	8.0×10^1
Gland Seal Steam Condenser	3.5×10^2	5.0×10^{-2}	1.7×10^1
Condensate Storage Tanks	1.6×10^3	5.0×10^{-2}	8.0×10^1
Low-Pressure Feedwater Heaters	7.5×10^4	5.0×10^{-3}	3.7×10^2
Evaporator Drain Tanks	1.0×10^1	5.0×10^{-2}	5.0×10^{-1}
Reheater Drain Tanks	8.4×10^2	5.0×10^{-2}	4.2×10^1
Moisture Separator Drain Tank	3.0×10^1	5.0×10^{-3}	1.5×10^{-1}
Main Turbine	2.6×10^3	5.0×10^{-4}	1.3×10^0
Steam Evaporator	2.0×10^3	5.0×10^{-3}	1.0×10^1
Turbine Bypass Valve Assembly	1.5×10^1	5.0×10^{-3}	7.5×10^{-1}
Moisture Separator Reheaters	1.8×10^4	5.0×10^{-3}	9.0×10^1
Seal Water Liquid Tank	1.2×10^1	5.0×10^{-2}	6.0×10^{-1}
Pumped Drain Tank	2.7×10^1	5.0×10^{-2}	1.4×10^0
High-Pressure Feedwater Heaters	1.7×10^4	5.0×10^{-3}	8.5×10^1
Total	2.0×10^5		1.2×10^3
Radwaste and Control Building			
Condensate Phase Separator Tanks	1.8×10^2	5.0×10^0	9.0×10^2
Condensate Backwash Receiver Tank	8.5×10^1	5.0×10^0	4.2×10^2
Waste Collector Tank	1.0×10^2	5.0×10^{-2}	5.0×10^0
Waste Surge Tank	1.9×10^2	5.0×10^0	9.5×10^2
Waste Sample Tanks	1.6×10^2	5.0×10^{-2}	8.0×10^0
Floor Drain Collector Tank	1.1×10^2	5.0×10^{-2}	5.5×10^0
Waste Sludge Phase Separator Tank	6.1×10^1	5.0×10^0	3.0×10^2
Floor Drain Sample Tank	7.8×10^1	5.0×10^{-2}	3.9×10^0
Chemical Waste Tanks	1.5×10^2	5.0×10^{-2}	7.5×10^0
Distillate Tanks	1.5×10^2	5.0×10^{-2}	7.5×10^0
Detergent Drain Tank	3.2×10^1	5.0×10^{-2}	1.6×10^1
Decontamination Solution Conc. Waste Tk.	2.3×10^1	5.0×10^0	1.2×10^2
Spent Resin Tank	1.3×10^1	5.0×10^0	6.5×10^1
Cleanup Phase Separator Tanks	6.8×10^1	5.0×10^0	3.4×10^2
Decontamination Solution Concentrator	1.9×10^1	5.0×10^0	9.5×10^1
Total	1.4×10^3		3.2×10^3

Source: NUREG/CR-0672 Vol. 2, Appendix E

[1] Note: 1 $Ci/m^2 = 2.2 \times 10^{10}$ dpm/100 cm^2

3.2.4 Levels of Internal Surface Contamination for Reference PWR

Radioactive contamination levels associated with internal surfaces of piping and equipment for a Reference PWR have been estimated in NUREG/CR-0130. At time of shutdown, the fractional contributions of various radionuclides deposited on internal surfaces of the primary loop of a PWR are shown in Table A3-17.

Estimates of internal surface deposition levels expressed in Ci/m^2 for major systems and components were based on models, which correlated external dose rate measurements with internal contamination analyses, taking into account source geometry and shielding factors (Table A3-18). Empirical dose rate measurements showed that reactor vessel and steam generator internal surfaces in contact with primary coolant, on average, would yield contamination levels of about 0.23 Ci/m^2 at time of shutdown.

Table A3-17. Estimates of Internal Surface Contaminants in a Reference PWR Primary System

Radionuclide	Deposited Radioactivity (µCi/m^2)	Fractional Radioactivity at Decay Times of:			
		Shutdown	10 Years	30 Years	50 Years
Cr-51	5.3×10^3	2.4×10^{-2}	—	—	—
Mn-54	8.0×10^3	3.6×10^{-2}	1.1×10^{-5}	—	—
Fe-59	1.8×10^3	8.2×10^{-3}	—	—	—
Co-58	1.0×10^5	4.6×10^{-1}	—	—	—
Co-60	7.1×10^4	3.2×10^{-1}	8.6×10^{-2}	6.2×10^{-3}	4.4×10^{-4}
Zr-95	8.8×10^3	5.6×10^{-2}	—	—	—
Nb-95	1.2×10^4	5.6×10^{-2}	—	—	—
Ru-103	5.9×10^3	2.6×10^{-2}	—	—	—
Cs-137	2.6×10^2	1.2×10^{-3}	9.5×10^{-4}	6.0×10^{-4}	3.8×10^{-4}
Ce-141	1.5×10^4	6.6×10^{-2}	—	—	—
TOTAL	2.3×10^5	1.0	8.7×10^{-2}	6.8×10^{-3}	8.2×10^{-4}

Table A3-18. Levels of Contamination and Estimated Quantities of Radioactive Corrosion Products Deposited on the Interior of PWR Reactor Systems

Systems	Surface (m^2)	Activity Level (Ci/m^2)	Total (Ci)
Reactor Vessel and Internals	5.7×10^2	0.23	130[1]
Steam Generators	1.9×10^4	0.23	4,400
Pressurizer	8.7×10^1	0.04	4
Piping (Except RCS)	1.1×10^3	0.06	60
RCS Piping	1.9×10^2	0.86	160
TOTALS	219×10^2		4,800

[1] Not included in this estimate is the very high activity from volumetrically distributed activation products

With a total surface area of 570 m^2, the reactor vessel and internal components were estimated to be contaminated with about 130 Ci. The four steam generators with a collective weight of 1,251 metric tons and combined surface area of about 19,000 m^2 were estimated to contain approximately 4,400 Ci or 90% of the total deposition level of activated corrosion products in the 89 metric ton pressurizer was assumed to be about 0.04 Ci/m^2. With an internal surface area of about 87 m^2, a total deposition of 4 Ci was estimated.

RCS piping includes those sections of piping interconnecting the reactor vessel, steam generators, reactor coolant pumps, and various other components as identified in Figure A3-1. RCS piping primarily involves large diameter thick-walled pipes. The inside diameter typically ranges from 699 mm to 787 mm with corresponding wall thickness between 59 and 66 mm. From dose rate measurements of about 600 mR/hr, the internal level of radioactivity for RCS piping was estimated at 0.86 Ci/m^2. For the internal surface area of about 190 m^2, the activity of 160 Ci was calculated for an estimated 100.8 metric tons of RCS piping.

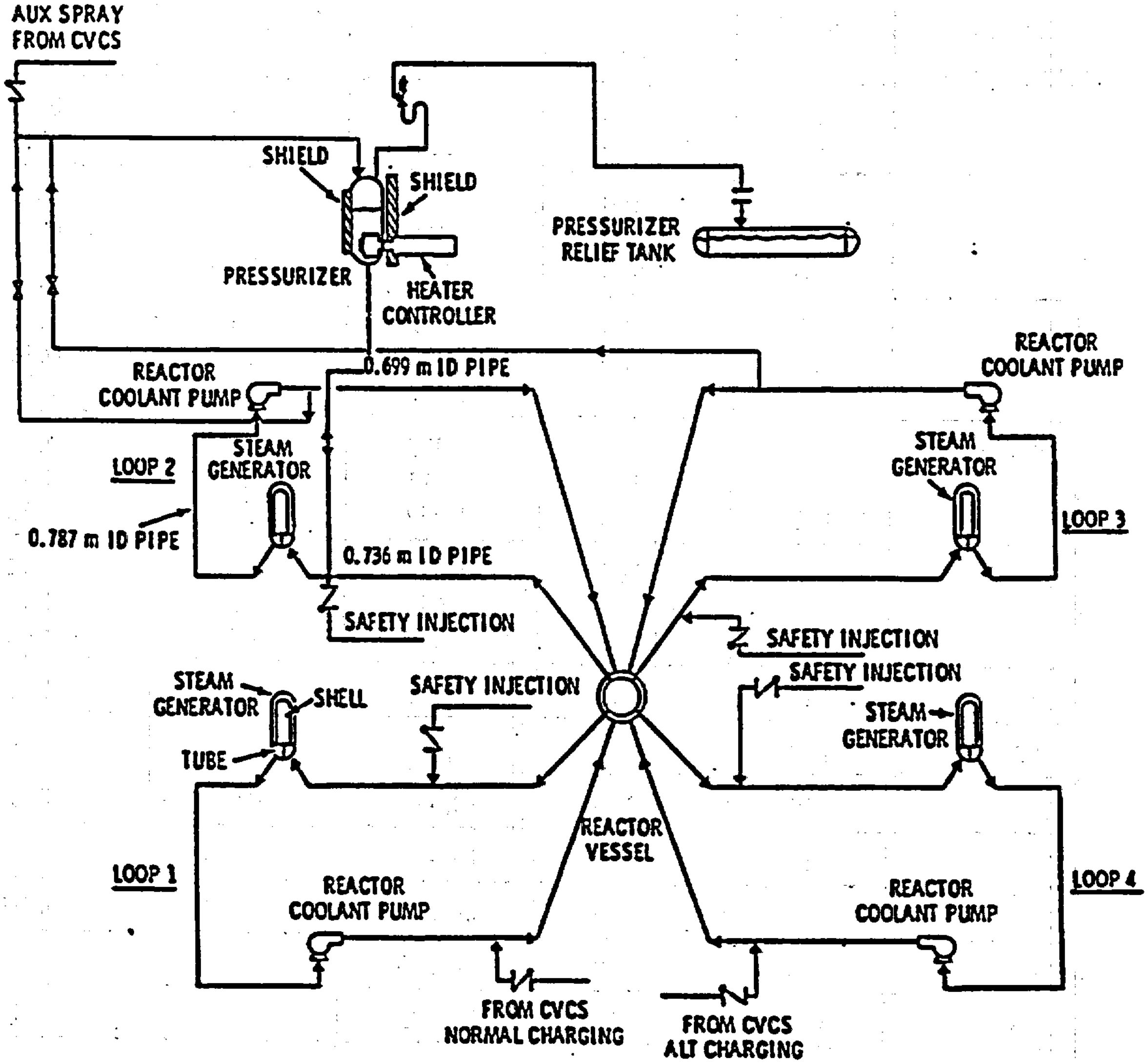

Figure A3-1. PWR Primary System Schematic and Piping Data

For non-RCS or auxiliary system piping, an average internal deposition of about 0.06 Ci/m^2 was derived based on external dose rate measurements. The collective inventory of 60 Ci for all non-RCS piping was estimated based on the piping quantities defined in Table A3-19.

Table A3-19. Non-RCS Contaminated Piping Data

Diameter Size (mm)	Schedule	Length (m)	Total Wt. (kg)	Total Inside Surface (m^2)	Total Activity (Ci)
12	80 160	120 120	198 238	5.3 4.1	0.32 0.25
19	40 80 160	240 360 570	205 400 1,675	8.0 10.8 28.3	0.48 0.65 1.70
25	40 80 160	60 180 420	152 590 1,800	5.1 14.0 22.7	0.31 0.84 1.36
25 - 38	40 80 160	120 330 540	493 1,811 3,967	15.7 40.1 58.6	0.94 2.41 3.52
51	40 80 160	300 480 1,050	1,655 3,642 11,850	50.3 75.4 143.6	3.02 4.52 8.62
71	160	140	2,985	29.4	1.76
102	160	180	6,128	50.2	3.01
152	160	300	20,972	128.7	7.72
203	160	140	15,924	70.7	4.24
254	140	365	29,750	134.1	8.05
305	140	90	18,370	74.0	4.44
356	140	100	25,475	92.3	5.54
Total			~148,000	~1,110	~60

3.3 Contamination of External Surfaces of Equipment and Structural Components

External surfaces of system components as well as floors, walls, and structural components become contaminated over the operating lifetime of a reactor plant from leaks and spills of radionuclides originating from reactor coolant water. While most liquid contamination remains highly localized to the vicinity of the leak/spill, some contamination may experience limited transfer through physical contact. More widespread contamination of external surfaces occurs when contaminants become airborne and passively settle out. Airborne contaminants are also the principal source of contamination of ducts, fans, filters, and other equipment that are part of the heating and ventilation and air conditioning systems (HVAC).

Mixtures of radionuclides typically found in primary coolant and their relative abundance in a PWR and BWR are given in Table A3-20 and Table A3-21, respectively.

Table A3-20. Radionuclides in Primary Coolant that Contribute to External Surface Contamination in the Reference PWR

Radionuclide	Half-Life (days)	Fractional Radioactivity at Decay Times of:			
		Shutdown	10 Years	30 Years	50 Years
Cr-51	2.8E+1	6.9E-4	—	—	—
Mn-54	3.1E+2	1.4E-3	4.2E-7	—	—
Fe-55	9.5E+2	2.2E-2	1.7E-3	9.9E-6	5.7E-8
Fe-59	4.5E+1	8.7E-4	—	—	—
Co-58	7.2E+1	7.5E-3	—	—	—
Co-60	1.9E+3	7.5E-2	2.0E-2	1.4E-3	1.0E-4
Sr-89	5.3E+1	1.2E-3	—	—	—
Sr-90	1.0E+4	6.9E-4	5.4E-4	3.4E-4	2.1E-4
Y-90	1.0E+4	6.9E-4	5.4E-4	3.4E-4	2.1E-4
Zr-95	6.5E+1	2.5E-4	—	—	—
Nb-95	3.5E+1	2.5E-4	—	—	—
Te-129m	3.4E+1	3.1E-4	—	—	—
I-131	8.0E+0	1.4E-2	—	—	—
Cs-134	7.5E+2	1.2E-1	4.1E-3	4.8E-6	5.4E-9
Cs-136	1.4E+1	1.1E-3	—	—	—
Cs-137	1.1E+4	7.5E-1	5.9E-1	3.7E-1	2.4E-1
TOTAL		1.0	6.2E-1	3.7E-1	2.4E-1

Table A3-21. Radionuclide Concentrations in Primary Coolant that Contribute to External Surface Contamination in Reference BWR

Radionuclide	Half-Life (days)	Concentration (µCi/g)	Fractional Radioactivity at Decay Times of:			
			Shutdown	10 Years	30 Years	50 Years
P-32	1.4E+1	2E-4	1.1E-3	—	—	—
Cr-51	2.8E+1	5E-3	5.3E-2	—	—	—
Mn-54	3.1E+2	6E-5	7.2E-4	2.3E-7	—	—
Fe-55	9.5E+2	1E-3	3.7E-1	2.5E-2	1.2E-4	6.0E-7
Fe-59	4.5E+1	3E-5	5.3E-4	—	—	—
Co-59	7.2E+1	2E-4	5.6E-3	—	—	—
Co-60	1.9E+3	4E-4	2.9E-1	7.8E-2	5.6E-3	4.0E-4
Ni-63	3.6E+4	1E-6	3.4E-3	3.2E-3	2.8E-3	2.4E-3
Zn-65	2.4E+2	2E-4	1.8E-2	4.5E-7	—	—
Sr-89	5.3E+1	1E-4	2.0E-3	—	—	—
Sr-90	1.0E+4	6E-6	1.5E-2	1.2E-2	7.0E-3	4.2E-3
Y-90	2.7E+0	6E-6	1.5E-2	1.2E-2	7.0E-3	4.2E-3
Y-91	5.9E+1	4E-5	8.1E-4	—	—	—
Zr-95	6.5E+1	7E-6	1.6E-4	—	—	—
Ru-103	4.0E+1	2E-5	2.9E-4	—	—	—
Ru-106	3.7E+2	3E-6	3.9E-4	—	—	—
Ag-110m	2.5E+2	1E-6	8.8E-6	3.2E-10	—	—
Te-129m	3.4E+1	4E-5	4.9E-4	—	—	—
I-131	8.0E+0	5E-3	1.5E-2	—	—	—
Cs-134	7.5E+2	3E-5	8.8E-3	3.1E-4	3.7E-7	4.4E-10
Cs-136	1.4E+1	2E-5	1.0E-4	—	—	—
Cs-137	1.1E+4	7E-5	1.8E-1	1.4E-1	9.0E-2	5.7E-2
Ba-140	1.3E+1	4E-4	2.0E-3	—	—	—
Ca-140	1.7E+0	4E-4	2.0E-3	—	—	—
Ce-141	3.2E+1	3E-5	3.4E-4	—	—	—
Ce-144	2.8E+2	3E-6	2.9E-4	3.1E-8	—	—
Pr-143	1.4E+1	4E-5	2.0E-4	—	—	—
Nd-147	1.1E+1	3E-6	1.2E-5	—	—	—
TOTAL		1.3E-2	1.0	2.7E-1	1.1E-1	6.8E-2

The amount of external surface contamination following 40 years of operation is likely to vary significantly among plants and is influenced by fuel integrity, primary coolant chemistry, operational factors, and reactor performance. In addition, a key operational factor is the effort at operating plants to clean up spills and to decontaminate accessible areas on a ongoing basis.

Although all nuclear utilities conduct routine radiological surveys that assess fixed and removable surface contamination, only limited data have been published in the open literature from which average contamination estimates can be derived. In this section, estimates of external surface contamination are provided that reflect (1) modeled data, (2) data published in the open literature, and (3) data submitted by individual utilities that have submitted a Decommission Plan.

3.3.1 Modeled Data for Reference Facility

Quantities and locations of structural surface contamination have been modeled for Reference PWR (NUREG/CR-0672). The model was based on an assumed release rate of one liter of primary coolant per day for 40 years. Deposition of contaminants on external surfaces was also correlated to ambient dose rates by means of the computer code ISOSHLD and placed in two discrete categories.

The first category is defined as low-level contamination areas with dose rates of 10 mR/hr in air at 1 meter from the surface. The second category was defined as areas of higher contamination with dose rates of 100 mR/hr in air at 1 meter from the surface. The structural surface contamination levels that correspond to dose rates of 10 and 100 mR/hr for the Reference BWR coolant water mixture were estimated to correspond to 2.5×10^{-3} Ci/m^2 and 2.5×10^{-2} Ci/m^2, respectively.

Table A3-22 summarizes the distribution and quantities of external surface contamination at shutdown. Surface contamination levels are expressed in units of disintegrations per minute per one-hundred square centimeters of surface (dpm/100 cm^2). The total deposited radioactivity on structural surfaces in the Reference BWR was estimated at 114 Ci.

Table A3-22. Surface Contamination Levels for Reference BWR at Shutdown

Building	Estimate Surface Area (m^2)	Radioactivity Deposited (Ci)	Avg. Contamination at Shutdown (dpm/100 cm^2)
Reactor Building	5145	74	3.16×10^8
(Contamination Level 1)(a)	(2403)	(5.7)	(5.22×10^7)
(Contamination Level 2)(b)	(2742)	(68.3)	(5.48×10^8)
Turbine Generator Bldg.	1817	4.4	5.33×10^7
(Contamination Level 1)(a)	(1767)	(3.2)	(3.98×10^7)
(Contamination Level 2)(b)	(50)	(1.2)	(5.28×10^8)
Radwaste & Control Bldg.	1953	35.8	4.03×10^8
(Contamination Level 1)(a)	(579)	(1.4)	(5.32×10^7)
(Contamination Level 2)(b)	(1374)	(34.4)	(5.51×10^8)
TOTAL		114 Ci	

(a) Contamination Level 1 corresponds to 2.5×10^{-3} Ci/m^2.

(b) Contamination Level 2 corresponds to 2.5×10^{-2} Ci/m^2.

Table A3-23 provides a more detailed breakdown of contamination levels by identifying major equipment/systems that are located within each of the aforementioned facility buildings.

Table A3-23. Estimated Structural Surface External Contamination in the Reference BWR[*]

Building/ Associated Equipment/System/Structure	Estimated Contaminated Surface Area (m^2)	Radioactivity Deposition Level	Deposited Radioactivity (Ci)
<u>Reactor Building</u>			
Containment Atmosphere Control	1.6×10^{1}	1	4.0×10^{-2}
Condensate (Nuclear Steam)	3.3×10^{1}	1	8.2×10^{-2}
Control Rod Drive	1.8×10^{2}	1	4.5×10^{-1}
Equipment Drain (Radioactive)	1.8×10^{1}	2	4.5×10^{-1}
Floor Drain (Radioactive)	7.4×10^{1}	2	1.8×10^{0}
Fuel Pool Cooling & Cleanup	1.2×10^{3}	1	3.0×10^{0}
Fuel Pool Cooling & Cleanup	2.8×10^{2}	2	7.0×10^{0}
High-Pressure Core Spray	1.1×10^{2}	1	2.7×10^{-1}
Low-Pressure Core Spray	1.4×10^{1}	1	3.5×10^{-2}
Main Steam	3.0×10^{2}	1	7.5×10^{-1}
Miscellaneous Wastes (Radioactive)	8.3×10^{1}	1	2.1×10^{-1}
Reactor Building Closed Cooling	1.2×10^{1}	1	3.0×10^{-2}
Reactor Core Isolation Cooling	1.5×10^{1}	1	3.8×10^{-2}
Reactor Water Cleanup	1.5×10^{2}	1	3.8×10^{-1}
Reactor Water Cleanup	1.7×10^{2}	2	4.2×10^{0}
Residual Heat Removal	1.7×10^{2}	1	4.2×10^{-1}
Standby Gas Treatment	4.0×10^{1}	1	1.0×10^{-1}
Traversing Incore Probe	8.0×10^{1}	1	2.0×10^{-1}
Primary Containment	2.2×10^{3}	2	5.5×10^{1}
Total			7.4×10^{1}

Table A3-23. Estimated Structural Surface External Contamination in Reference BWR[*] (Continued)

Building/ Associated Equipment/System/Structure	Estimated Contaminated Surface Area (m²)	Radioactivity Deposition Level	Deposited Radioactivity (Ci)
Turbine Generator Building			
Air Removal	3.9×10^1	1	9.7×10^{-2}
Condensate (Nuclear Steam)	6.6×10^2	1	1.6×10^{-1}
Condenser Off Gas Treatment	1.8×10^2	1	4.5×10^{-1}
Equipment Drain (Radioactive)	2.5×10^1	2	6.2×10^{-1}
Floor Drain (Radioactive)	2.5×10^1	2	6.2×10^{-1}
Heater Drain	9.1×10^1	1	2.3×10^{-1}
Main Steam	1.7×10^2	1	4.2×10^{-1}
Miscellaneous Drain & Vent	1.9×10^1	1	4.7×10^{-2}
Reactor Feedwater	6.9×10^2	1	1.7×10^0
Miscellaneous Wastes (Radioactive)	9.0×10^0	1	2.2×10^{-2}
Total			4.4×10^0
Radwaste and Control Building			
Condensate Filter Demineralizer	3.6×10^2	2	9.0×10^0
Condenser Off Gas Treatment	3.2×10^2	1	8.0×10^{-1}
Equipment Drain (Radioactive)	4.3×10^1	1	1.1×10^{-1}
Equipment Drain (Radioactive)	1.8×10^2	2	4.5×10^0
Floor Drain (Radioactive)	1.2×10^1	1	3.0×10^{-2}
Floor Drain (Radioactive)	1.9×10^2	2	4.8×10^0
Floor Pool Cooling & Cleanup	5.4×10^1	2	1.4×10^0
Miscellaneous Wastes (Radioactive)	2.4×10^1	1	6.0×10^{-2}
Miscellaneous Wastes (Radioactive)	1.9×10^2	2	4.8×10^0
Process Waste (Radioactive)	1.8×10^2	1	4.5×10^{-1}
Process Waste (Radioactive)	2.7×10^2	2	6.7×10^0
Reactor Water Cleanup	1.3×10^2	2	3.2×10^0
Total			3.6×10^1

Estimated total deposited radioactivity on contaminated external surfaces = 1.14×10^2 Ci

[*] Source: NUREG/CR-0672

Modeled Estimates Versus Empirical Study Data. External surface contamination corresponding to Level 1 (2.5×10^{-3} Ci/m^2 or 5.2×10^7 dpm/100 cm^2) and Level 2 (2.5×10^{-2} Ci/m^2 or 5.5×10^8 dpm/100 cm^2) are not uncommon and have been observed in most reactor facilities. Table A3-24 presents study data that focused on the most highly contaminated surfaces at six nuclear power plants (NUREG/CR-4289). Contamination levels corresponding to modeled values (i.e., Level 1 and Level 2), however, were restricted to small areas that had experienced spills, leaks, or intense maintenance, such as the reactor sump area, RCS coolant pumps, and radwaste system components. The study data also showed that when surfaces were coated with sealant or epoxy paint, nearly all contamination resided on or within the surficial coating and was readily removable.

Table A3-24. Ranges of Radionuclide Associated with Highly Contaminated External Surfaces at Six Nuclear Generating Stations

Radionuclide	Half-life (yr)	Concentration Range (pCi/cm^2)	Average Concentration (dpm/100 cm^2)	
Co-60	5.27	590 - 460,000	2.4×10^7	(5)*
Ni-59	75,000	30 - 2,400	1.9×10^5	(3)*
Ni-63	100	3,100 - 6,400	1.0×10^6	(2)*
Sr-90	28.5	1.6 - 480	3.7×10^4	(4)*
Tc-99	2.13E+5	0.27 - 2.4	3.5×10^2	(3)*
Cs-137	30.2	550 - 2.0E+6	8.1×10^7	(6)*
Eu-152	12.4	9 - 3,100	2.2×10^5	(3)*
Eu-154	8.5	90 - 1,500	1.5×10^5	(3)*
Eu-155	4.96	10 - 500	1.3×10^4	(2)*
Pu-238	87.8	0.025 - 48	3.1×10^3	(4)*
Pu-239, 240	24,400	0.089 - 21	1.7×10^3	(4)*
Am-241	433	0.10 - 30	1.9×10^3	(4)*
Cm-244	18.1	0.013 - 0.026	3.5×10^0	(3)*

* Number of reactor units included to calculate the average value.

In summary, the modeled external surface contamination levels cited in NUREG/CR-0672 for Reference BWR appear excessive in terms of their projected surface areas and total plant inventory. The primary model parameter regarding the release of one liter of primary coolant per day that is allowed to buildup over a forty-year period of plant operation is not only without technical basis but ignores the ongoing decontamination efforts that exist at all nuclear facilities. For these reasons, modeled data contained in NUREG/CR-0672 are not considered suitable.

3.3.2 Surface Contamination Levels Reported by Facilities Preparing for Decontamination and Decommissioning (D&D)

PWR. By coincidence (as was previously acknowledged), the Trojan Nuclear Plant (TNP), which had served as the Reference PWR facility in the 1978 study (NUREG/CR-0130), has been permanently shutdown and has submitted a Decommissioning Plan. External surface contamination inventories at this facility are summarized in TNP's Decommissioning Plan and have been reproduced in Table A3-25. Estimates were based on historical survey data and recent structural surveys performed in support of the Decommissioning Plan's required Radiological Site Characterization.

Table A3-25. Inventory of External Surface Contamination at Trojan Nuclear Plant

Location	Activity (Ci)
Structures	
Containment Building	0.024
Auxiliary Building	0.002
Fuel Building	0.001
Main Steam Support Structure	0.001
Turbine Building	0.002
Total	0.030

Combined contamination inventories for the containment building, auxiliary building, fuel building, and the main steam support structure are estimated at 0.03 Ci. Note that this value is more than three orders of magnitude lower than the modeled estimate for the Reference BWR.

More detailed contamination data relating to external surfaces at TNP were recently cited in a draft report issued by the NRC (Draft NUREG-1496). Survey data primarily defined removable floor contamination levels obtained by smears. However, such measurements may reasonably be assumed to also represent metal surfaces of reactor systems and structural metal components.

A summary of removable external surface contamination levels at TNP are given in Table A3-26.

Table A3-26. Areal Surface Contamination Levels Based on Survey Measurements at TNP Preparing for D&D[*]

Building	Approximate Floor Surface Area, m^2	Estimated % of Floor Area Contaminated	Estimated % Needing Cleanup, m^2	Removable Measured Contamination Level, dpm/100 cm^2
Reactor Containment	1,900	100	1,900	1,100 - 55,000
Auxiliary (6 levels)	4,000	1 - 5	40 - 200	< 1,100 - 7,900
Fuel Building (5 levels)	5,000	1 - 5	50 - 250	< 1,100 - 5,000
Turbine Building	5,700 per level	<< 1	~ 0	< 1,000
Control Building	700 per level	<< 1	~ 0	< 1,000

[*] Source: NUREG-1496

The <u>auxiliary</u> and <u>fuel buildings</u> also exhibited some areas of floor contamination, but not to the extent of that observed in the reactor containment building. Based on survey reports, about 1 to 5% of the floor area (representing about 40 to 200 m^2) in the auxiliary building has radioactive contamination levels in the range of 1,100 to 7,900 dpm/100 cm^2. The fuel handling building also has a small amount of floor contamination, estimated at approximately 50 to 250 m^2, with contamination levels ranging from about 1,100 to 5,000 dpm/100 cm^2.

Other buildings, including the turbine building and the control building, did not have measurable contamination on any surfaces.

It is important to note, however, that the quantitative estimates in Table A3-25 reflect contamination that is <u>removable</u> (i.e., by wiping a 100 cm^2 area with a dry filter paper).

Reasonable estimates of total surficial contamination levels (i.e., fixed and removable) may be obtained by multiplying values in Column 5 of Table A3-26 by a factor whose value may range from 5 to 10.

BWR. Values similar to those reported in the TNP's Decommissioning Plan have also been reported in the decommissioning plan submitted for Humboldt Bay Unit 3. Excerpts of survey measurements (as they appear in the D-Plan) are contained in Addendum #2. Horizontal surfaces (i.e., floors) exhibited contamination levels that on average were about one order of magnitude higher than vertical surfaces (i.e., walls) with values ranging from below detection limits up to several million dpm per 100 cm^2 for select floor areas (e.g., under the reactor vessel). When relatively small areas of high contamination are excluded, average external surface contamination was generally between 5,000 dpm/100 cm^2 to 100,000 dpm/100 cm^2.

From the above-cited data, it is concluded that, within the common variabilities of contamination levels in nuclear plants, the survey data reported in decommissioning plans for the Trojan and Humboldt Bay facilities provide a reasonable estimate of surface contamination levels for other PWRs and BWRs.

4.0 BASELINE METAL INVENTORIES

4.1 Baseline Metal Inventories for Reference PWR

The total amounts of metals contained in significant quantities in a typical 1,000 MWe pressurized water reactor (PWR) power plant have been quantified in a 1974 study of material resource use and recovery in nuclear power plants (Bryan and Dudley 1974). Material estimates were made using various methods that included: (1) amounts of raw materials purchased for construction (e.g., reinforcing steel and structural steel required for construction), (2) weights of materials contained in equipment and machinery based on manufacturers' specifications and technical journals (e.g., determination of carbon steel, stainless steel, copper and other metals in electric motors); and (3) the U.S. Atomic Energy Commission facility accounting system, which identified individual items.

Summary estimates of composite materials used to construct a 1971-vintage 1,000 MWe PWR power plant are given in Table A4-1.

Table A4-1. Inventory Estimates of Materials Used to Construct a 1971-Vintage, 1,000 MWe, Pressurized Water Reactor Facility

Material	Total Quantity (Metric tons)
Metals:	
Carbon Steel	3.3×10^4
(Rebar)	(1.3×10^4)
(All Other)	(2.0×10^4)
Stainless Steel	2.1×10^3
Galvanized Iron	1.3×10^3
Copper	6.9×10^2
Inconel	1.2×10^2
Lead	4.6×10^1
Bronze	2.5×10^1
Aluminum	1.8×10^1
Brass	1.0×10^1
Nickel	1.0
Silver	< 1.0

Source: Bryan and Dudley 1974

Carbon steel is by far the most abundant metal used in the construction of a nuclear power plant. It is used in piping and system components when the need for corrosion resistant stainless steel is not of significant importance. A large percentage is also used in structural components that include rebar, I-beams, plates, grates, and staircases. A breakdown of material quantities used in reactor plant structures and plant systems is provided in Table A4-2. Essentially 50% or 16,519 metric tons out of total of 32,731 metric tons of carbon steel is used for structural components with the other one-half used in plant equipment. Of the more than 16,000 metric tons of carbon steel employed in plant equipment/systems, about two-thirds (i.e., 10,958 metric tons) are contained in turbine plant equipment. Barring significant leakage in steam generators, equipment in this grouping as well as electric plant equipment, equipment identified as "miscellaneous," and "structures" are not likely to be exposed to radionuclides/radioactivity and are, therefore, not likely to contribute significant quantities of scrap metal.

The primary sources of contaminated scrap metal in a PWR are identified as shaded areas in Table A4-2 and involve all items associated with Reactor Plant Equipment with additional quantities contributed by "Fuel Storage," select structural components, HVAC systems, and other items that are identified in detail in Section 5.0 below.

Inspection of Table A4-2 also reveals that the use of corrosion resistant stainless steel is almost totally confined to reactor plant and turbine plant systems. Of the total 2,080 metric tons of stainless steel, essentially all of the 1,154.6 metric tons associated with reactor plant systems and the 21.1 metric tons that line the fuel pool can be assumed contaminated.

4.2 Baseline Inventories for Reference BWR

Inventories for a 1,000-MWe BWR reference plant have been estimated by adjusting Bryan and Dudley's 1974 Reference PWR plant data taking into account BWR characteristics (NUREG/CR-0672).

Table A4-2. Breakdown of Material Quantities

Metric Tons[1,2]

System	Carbon Steel	Stainless Steel	Galvanized Iron	Copper	Inconel	Lead	Bronze	Aluminum	Brass	Nickel	Silver
Structures/Site	16519.3	28.6	814.2	33.1	0.0	33.1	0.2	1.2	2.9	0.1	0.1
Site Improvements	1692.9	0.0	17.9	1.5	0.0	0.7	0.0	0.1	0.0	0.0	0.0
Reactor Building	7264.2	5.7	301.2	9.3	0.0	0.0	0.0	0.1	0.3	0.0	0.0
Turbine Building	3641.2	0.0	196.4	1.6	0.0	0.0	0.1	0.8	1.4	0.0	0.0
Intake/Discharge	333.7	0.0	3.6	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Reactor Auxiliaries	1358.7	0.0	109.8	0.8	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Fuel Storage	364.6	21.1	43.4	0.3	0.0	0.0	0.0	0.1	0.1	0.0	0.0
Miscellaneous Bldgs.	1864	1.8	141.9	19.4	0.0	32.4	0.1	0.1	0.9	0.1	0.1
Reactor Plant Equipment	334.9	1154.6	5.5	50.4	124.2	4.5	0.5	5.2	0.0	0.0	0.0
Reactor Equipment	430.0	275.1	0.0	6.8	124.1	0.0	0.0	0.0	0.0	0.0	0.0
Main Heat Trans. System	1686.5	202.5	1.6	9.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Safeguards Cool. System	274.2	199.1	1.1	2.9	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Radwaste System	35.2	31.9	0.8	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fuel Handling System	82.0	67.0	0.3	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other Reactor Equipment	823.5	230.3	1.7	1.5	0.0	4.5	0.4	0.0	0.0	0.0	0.0
Instrumentation & Control	113.5	148.7	0.0	29.0		0.0	0.0	5.2	0.0	0.0	
Turbine Plant Equipment	10,958.3	883.2	4.7	51.4	0.0	0.0	21.5	1.2	6.9	0.0	0.0
Turbine-Generator	4138.6	129.9	0.5	35.2	0.0	0.0	19.7	0.0	0.0	0.0	0.0
Heat Rejection Systems	2501.1	9.1	2.2	3.0	0.0	0.0	0.7	0.0	0.4	0.0	0.0
Condensing Systems	1359.8	392.3	0.6	1.3	0.0	0.0	0.3	0.0	1.5	0.0	0.0
Feed-Heating System	1367.7	221.2	0.5	1.2	0.0	0.0	0.3	0.0	3.9	0.0	0.0
Other Equipment	1541.3	89.4	0.9	0.7	0.0	0.0	0.5	0.0	1.1	0.0	0.0
Instrumentation & Control	49.8	41.3	0.0	10.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0
Electric Plant Equipment	965.5	0.0	431.0	556.5	0.0	6.8	2.5	4.1	0.0	0.6	0.4
Switchgear	30.4	0.0	1.4	2.8	0.0	0.0	0.7	0.0	0.0	0.0	0.3
Station Service Equip.	654.1	0.0	8.5	19.0	0.0	6.8	0.7	0.0	0.0	0.0	0.1
Switchboards	87.0	0.0	0.0	13.5	0.0	0.0	0.1	4.1	0.0	0.0	0.0
Protective Equipment	5.9	0.0	0.0	39.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
Structures & Enclosure	112.5	0.0	421.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Power & Control Wiring	75.6	0.0	0.0	482.2	0.0	0.0	0.5	0.0	0.0	0.6	0.0
Miscellaneous Equipment	843.2	13.7	2.0	2.6	0.0	2.0	0.4	6.5	0.3	0.0	0.0
Transportation & Lifting	529.3	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Air & Water Service Sys.	232.5	6.0	0.0	1.1	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Communications Equip.	4.7	0.0	0.6	1.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
Furnishings & Fixtures	76.7	7.7	1.4	0.0	0.0	2.0	0.4	6.1	0.0	0.0	0.0
ENTIRE PLANT	32,731.2	2080.1	1257.4	694.0	124.2	46.4	25.1	18.2	10.1	0.7	0.5

[1] Source: Bryan and Dudley 1974

[2] Shaded areas identify PWR equipment/systems with significant amounts of radioactive contamination.

With regard to plant inventories of steel, notable differences between a PWR and BWR are that the latter has less heat-transfer piping and the absence of steam generators, but more ex-vessel primary components inclusive of a pressure suppression chamber. An additional significant difference between the PWR and BWR is the estimated quantity of rebar used for concrete reinforcement. Of the 32,700 tons total in the Reference 1,000 MWe PWR, Bryan and Dudley estimated that about 13,300 tons is rebar; for the 1,000 MWe Reference BWR, the collective weight of rebar was estimated at 18,300 tons (NUREG/CR-0672).

Beyond an increase in total steel required to construct a BWR, a second major difference is the enhanced percentage of steel (and other metals) that is contaminated. This is due to the fact that, under normal operating conditions of a BWR, radionuclides enter the steam flow and contaminate turbine plant equipment that in a PWR may generally be assumed uncontaminated.

Table A4-3 identifies material estimates for a 1,000-Mwe BWR plant. Material estimates for metals other than carbon and stainless steel for the 1,000-MWe Reference BWR are assumed to be identical to those of the 1,000-MWe Reference PWR.

Table A4-3. Inventory Estimates of Materials Used to Construct a 1,000-Mwe Boiling Water Reactor Facility

Material	Total Quantity (Metric tons)
Metals:	
Carbon Steel	3.4×10^4
(Rebar)	(1.8×10^4)
(All Other)	(1.6×10^4)
Stainless Steel	2.1×10^3
Galvanized Iron	1.3×10^3
Copper	6.9×10^2
Inconel	1.2×10^2
Lead	4.6×10^1
Bronze	2.5×10^1
Aluminum	1.8×10^1
Brass	1.0×10^1
Nickel	1.0
Silver	< 1.0

Source: NUREG/CR-0672

4.3 The Applicability of Reference Facility Data to the Nuclear Industry

The applicability of material estimates cited by Bryan and Dudley (1974) to all currently licensed U.S. facilities is not without some difficulty. The current U.S. nuclear power plant inventory of 123 units is composed of not only different designs but also highly variable power ratings.

Reactor power plant designs reflect evolving standards over four decades for plant safety and environmental concerns. For example, Bryan and Dudley's reference plant used run-of-river cooling, which is not applicable to more recent nuclear facilities that employ cooling towers of various designs, holding ponds, sprays, etc. Significant quantities of materials are involved in some of these alternative cooling systems. Additionally, the 1979 accident at the Three Mile Island facility mandated revised safety standards, which have added to the material inventory of more recent nuclear plants.

Material inventories that reflect evolving changes in plant design, however, have not been adequately addressed in the open literature. Adjustments to material inventories for individual facilities will, therefore, be limited to the reactor's power rating by means of a scaling factor.

Scaling Factors. In general, it is reasonable to assume a positive correlation between a plant's power rating and its material inventory. In a recent draft report prepared by Argonne National Laboratory for the U.S. Department of Energy, a scaling method was employed that was based on PWR and BWR vessel mass data (Nuclear Engineering International data 1991, 1992, and 1993). In these reports, it is assumed that all metal inventories for both PWRs and BWRs correlate to the corresponding reference plant in proportion to the design power rating to the 2/3 power (i.e., $MWe^{2/3}$). Thus, the scaling factor of 0.86 and 1.13 would correspond to plants with an 800 MWe and 1,200 MWe generating capacity, respectively.

This scaling factor was applied multiplicatively to each of the 123 plants (identified in Addendum #1) for estimating total industry inventories (Table A4-4). For most metal groupings, the all-inclusive inventory estimates include a majority component that is not radiologically contaminated.

Table A4-4. Summary of Total Metal Inventories Potentially Available for Recycling (Metric Tons)

	Reference		Industry		
	BWR	PWR	All BWRs	All PWRs	Total
Carbon Steel (Rebar) (All Other)	3.4×10^4 (1.8×10^4) (1.6×10^4)	3.3×10^4 (1.3×10^4) (2.0×10^4)	1.1×10^6 (6.0×10^5) (5.4×10^5)	2.5×10^6 (1.0×10^6) (1.5×10^6)	3.6×10^6 (1.6×10^6) (2.0×10^6)
Stainless Steel	2.1×10^3	2.1×10^3	7.1×10^4	1.6×10^5	2.3×10^5
Galvanized Iron	1.3×10^3	1.3×10^3	4.4×10^4	1.0×10^5	1.4×10^5
Copper	6.9×10^2	6.9×10^2	2.3×10^4	5.4×10^4	7.7×10^4
Inconel	1.2×10^2	1.2×10^2	4.0×10^3	9.4×10^3	1.3×10^4
Lead	4.6×10^1	4.6×10^1	1.6×10^3	3.6×10^3	5.2×10^3
Bronze	2.5×10^1	2.5×10^1	8.4×10^2	2.0×10^3	2.8×10^3
Aluminum	1.8×10^1	1.8×10^1	6.1×10^2	1.4×10^3	2.0×10^3
Brass	1.0×10^1	1.0×10^1	3.4×10^2	7.8×10^2	1.1×10^3
Nickel	1.0	1.0	3.4×10^1	7.8×10^1	1.1×10^2
Silver	< 1.0	< 1.0	$< 3.4 \times 10^1$	$< 7.8 \times 10^1$	$< 1.1 \times 10^2$

APPENDIX B

RECYCLING OF ALUMINUM SCRAP

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RECYCLING OF ALUMINUM SCRAP

1.0 INTRODUCTION

To assess future potential impacts of recycling aluminum scrap metal from nuclear facilities, it is critical to evaluate current recycling processes. It is also important to have an understanding of the aluminum industry and the quantitative and dynamic relationship between mined ore and the recycling of aluminum and aluminum alloys. This report presents an overview of the aluminum industry and describes common processes or techniques used in the reclamation of aluminum scrap.

Aluminum is the second most abundant metallic element in the Earth's crust after silicon, yet it has only been produced in commercial quantities for just over 100 years. It weighs about one-third as much as steel or copper; is malleable, ductile, and easily machined and cast; and has excellent corrosion resistance and durability. Measured in either quantity or value, aluminum's use exceeds that of any other metal except iron, and it is important in virtually all segments of the world economy.

1.1 Aluminum Uses

Aluminum is used in a wide range of products such as beverage cans, foil wrap, automobiles, airplanes, trucks, windows, doors, aluminum siding, mobile homes, bridges, street signs, wiring household appliances, outdoor furniture, electronic devices, etc. These end uses for aluminum metal are usually divided into seven major categories or industries: containers and packaging, transportation, building and construction, electrical, consumer durables, machinery and equipment, and other miscellaneous uses.

In 1994, the U.S. Bureau of Mines (USBM) reported that the transportation industry became the largest consumer of aluminum products, representing 24.7 percent of the total consumption, closely followed by the container and packaging industry with 24.4 percent (see Table B-1). Prior to 1994, the container and packaging industry had dominated the U.S. aluminum market since the mid-1970's. The use of aluminum by the transportation industry, especially the automotive sector, has steadily grown over the last several years; and based on announcements of new aluminum applications by the automotive industry, this growth is expected to continue into the future.

Table B-1. Distribution of End-Use Shipment of Aluminum Products in the United States, By Industry

Industry	1993 Quantity (1,000 metric tons)	1993 Percent of Grand Total	1994 Quantity (1,000 metric tons)	1994 Percent of Grand Total
Containers and packaging	2,180	26.0	2,280	24.4
Transportation	1,970	23.5	2,310	24.7
Building and construction	1,240	14.7	1,400	15.0
Electrical	609	7.3	677	7.2
Consumer durables	563	6.7	647	6.9
Machinery and equipment	477	5.7	572	6.1
Other markets	259	3.1	276	2.9
Total to domestic users	7,300	87.0	8,160	87.2
Exports	1,090*	13.0	1,200*	12.8
Grand Total	8,390	100.0	9,360	100.0

* Estimated values.
Source: USBM 1995

1.2 Aluminum Production

1.2.1 Primary Production

"Primary" aluminum, or virgin metal, is produced from bauxite ore. The bauxite is refined to remove impurities, such as iron oxide. This multi-stage refining process produces a fine, white powder called alumina, which is a compound composed of aluminum and oxygen. The alumina is shipped to a reduction plant, or smelter, where it is continuously fed into pots that dissolve the alumina into a molten salt. An electrolytic process, or the passing of direct electric current through the pot, is then used to separate the aluminum and oxygen. When current is applied, the molten aluminum settles to the bottom of the pot. The molten aluminum, which is 99.5 percent pure, is either transported to a holding furnace, where it can be poured into the various forms of ingots, or it may be alloyed (other metals added) to produce desired composition and characteristics of the final product. (Note: Currently, aluminum scrap is <u>not</u> a raw material used in the primary aluminum production process.)

In 1994, the world production of primary aluminum (produced by 43 countries) was approximately 19.1 million metric tons. At 3.3 million metric tons, the United States was the largest producer with 17 percent of the world total, followed by Russia with 14 percent, and Canada with 12 percent (USBM 1995).

Nevertheless, domestic primary aluminum production decreased significantly in 1994 to its lowest level in 7 years. Thirteen companies operated 22 primary aluminum reduction plants and 1 plant remained temporarily closed. Montana, Oregon, and Washington accounted for 36 percent of the production; Kentucky, North Carolina, South Carolina, and Tennessee, 20 percent; and other States, 44 percent (USBM 1995). The principal cause for reduced primary production in recent years is the steady increase in quantities of recycled aluminum scrap as discussed below.

1.2.2 Secondary Recovery

Secondary aluminum, or the recovery of aluminum from scrap, has become an important component of the supply/demand relationship in the United States. The industry's recycling operations, commonly referred to as the "secondary aluminum industry," uses purchased scrap as its "raw" material. Purchased aluminum scrap is classified as "new" (manufacturing) scrap and "old" scrap (discarded aluminum products).

In 1994, metal recovered from both new and old scrap reached an historic high of approximately 3.1 million tons, according to data derived by the U.S. Bureau of Mines from its "Aluminum Scrap" survey of 95 U.S. companies and/or plants. Fifty-one percent of this recovered metal came from new scrap and 49 percent from old scrap. The predominant type of purchased scrap was aluminum used beverage container (UBC) scrap, accounting for more than one-half of the old scrap consumed.

According to figures released by the Aluminum Associates, Inc., the Can Manufacturers Institute, and the Institute of Scrap Recycling Industries, Inc. a record 64.7 billion aluminum cans were recycled in the United States during 1994. The recycling rate, based on the number of cans shipped during the year, increased to 65.4 percent.

1.3 Summary of Salient Statistics

Table B-2 provides a summary of salient statistics regarding U.S. production, consumption, and foreign trade of aluminum for the period of 1990 through 1994. As is evident in this table, the decline in primary aluminum production throughout the period was offset by the rise in secondary recovery of the metal. The secondary aluminum industry presently contributes nearly one-half of the total domestic supply.

Table B-2. Primary Aluminum Production Versus Secondary Recovery in the U.S.
(Thousand Metric Tons)

	1990	1991	1992	1993	1994
Primary Production	4,048	4,121	4,042	3,695	3,299
Secondary Recovery	2,390	2,290	2,760	2,940	3,080
New Scrap	1,030	969	1,140	1,310	1,580
Old Scrap	1,360	1,320	1,610	1,630	1,500
Inventories:					
Aluminum Industry	1,820	1,780	1,880	1,980	2,070
LME* Stocks in U.S. Warehouse	—	168	214	168	16
National Defense Stockpile	2	2	57	57	57
Exports (crude & semicrude)	1,660	1,760	1,450	1,210	1,1370
Imports for consumption	1,510	1,490	1,730	2,540	3,380
Consumption, apparent**	5,260	5,040	5,730	6,600	6,880

* London Metal Exchange

** Defined as domestic primary metal production + secondary recovery + imports - exports + adjustments for Government and industry stock changes - recovery from purchased new scrap.

Source: USBM 1995

2.0 ALUMINUM RECYCLING

2.1 Overview

The aluminum recycling industry has developed into a major market force in the domestic aluminum industry. Recycling of scrap provides a source of aluminum that not only helps the aluminum industry maintain its growth but also helps conserve energy and slow the depletion of bauxite resources.

Aluminum is a recyclable material of significant economic importance. The U.S. aluminum industry consumes 1 percent of the nation's energy, largely in the form of electricity. The amount of electricity used per pound of metal in smelting, which accounts for about two-thirds of the industry's total energy consumption, has decreased steadily, and today the average is about 7 kW h. Aluminum can be recycled for less than 5 percent of the energy required for producing virgin metal.

Aluminum recovered from scrap has shown a tenfold increase since 1950. The recovery of aluminum from old scrap has shown an even more rapid expansion over the same period of time. Increased costs for energy and growing concerns over waste management have provided the impetus for increase recycling rates. Improvements in recycling technologies and changes in the end-use consumption patterns have also contributed to the increase in aluminum scrap recovery.

2.2 Aluminum Scrap Supply Stream

Aluminum scrap enters the supply stream of the secondary aluminum industry through two major, broadly classified sources: (1) new scrap, generated by the fabrication of aluminum products, and (2) old scrap, which becomes available when consumer products have reached the end of their economic life and have been discarded. New scrap includes solids, such as new casting scrap, clippings or cuttings of new sheet, rod, wire, and cable, borings and turnings from machinery operations; residues (e.g. drosses, skimmings, spillings, and sweepings); and surplus products (mill products and castings). Old scrap includes products such as automobiles, aluminum windows/doors/siding, used beverage cans, and cooking utensils. Obsolete industrial products, such as transmission cables, aircraft, and other similar items, outdated inventory materials, production overruns, out-of-specification products, etc., are also classified as old scrap.

The new scrap may be purchased directly from the manufacturer, i.e., machining chips from automobile producers and machine shops, or from scrap dealers that have consolidated the scrap material from many manufacturers.

Old scrap is purchased from dealers who obtain the material from industrial sources and scrap collectors and prepare the various materials for resale. The dealer operates a collection and process facility where the aluminum scrap is segregated by type, cleaned to a condition making it commercially usable, and packaged by baling and other means for shipment in carload or truckload quantities to an industrial consumer. Purchased scrap may be in its original form or it may have already been melted in sweat furnaces and formed into a pig or ingot (termed sweated pig) or 1000 pound sows. Sweat furnaces are used to separate iron from old cast products since iron, in high concentrations, is detrimental to mechanical properties of most aluminum alloys. Sweating consists of placing the scrap metal on a sloping hearth or grate in a furnace and raising the temperature to about 1400°F. At this temperature, the aluminum melts, runs off, and is collected, leaving the iron behind.

An overview of the production/use/reclamation pathways and cycles of virgin aluminum, new aluminum scrap, and old aluminum scrap are depicted schematically in Figure B-1.

2.3 Classifications of Aluminum Alloys

Controlling the composition of aluminum recovered from scrap is essential to producing marketable secondary alloys. Therefore, it is necessary to provide a brief description of the aluminum alloys that are found in the solid waste stream. Aluminum alloys are divided into two distinct categories according to how they are formed (i.e., cast alloys and wrought alloys). Cast alloys are those specially formulated to flow into a sand or permanent mold, to be die cast, or to be cast by any other process where the casting is the final form. Wrought alloys are alloys that have been mechanically worked after casting. The "wrought" category is broad, since aluminum can be formed by virtually every known process: Wrought forms include sheet and plate, foil, extrusions, bar and rod, wire, forgings and impacts, crown or extruded tubing, and others.

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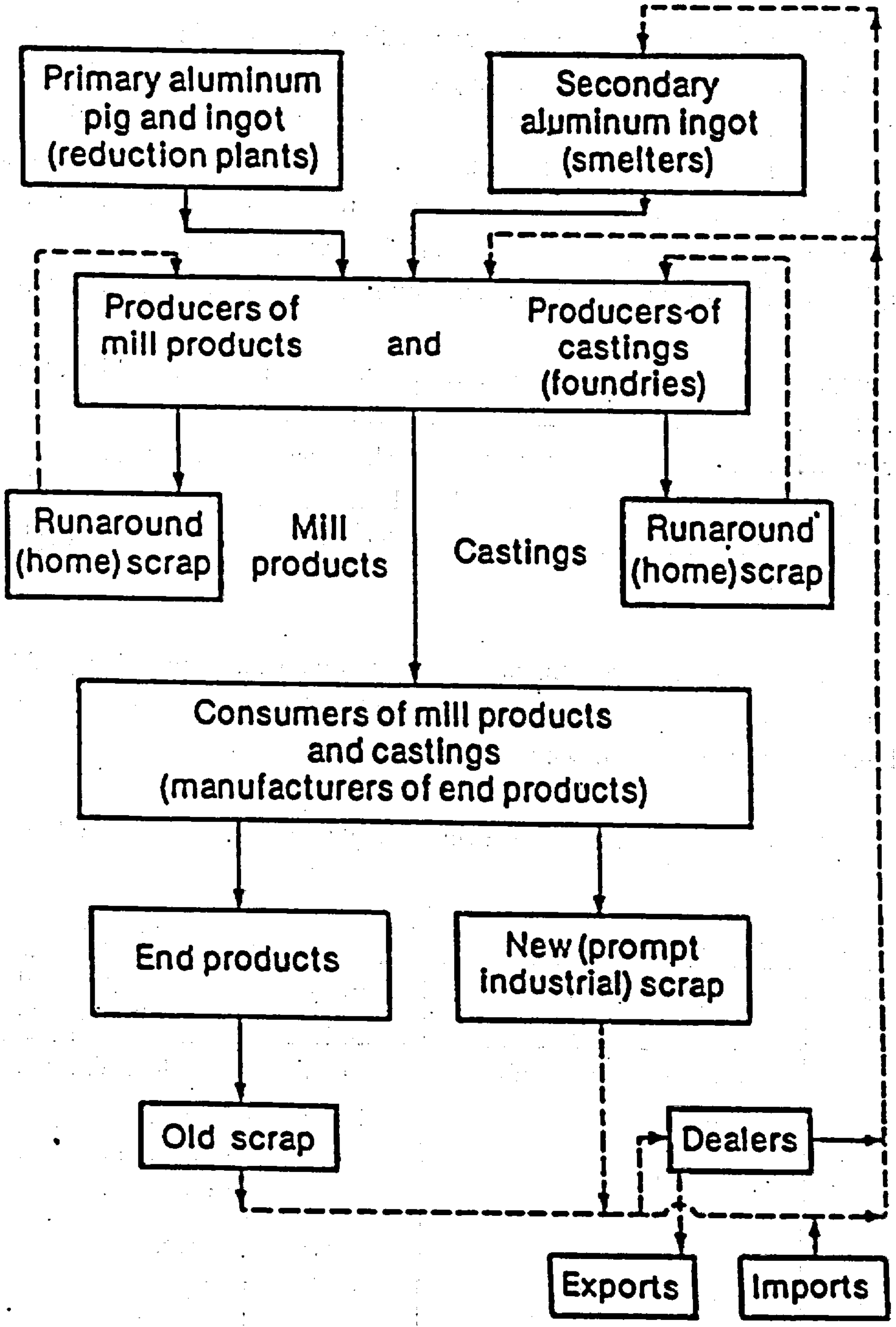


Figure B-1. Pathways and Cycles of Aluminum Usage.
(Source: Aluminum Recycling Casebook, 1985)

Aluminum casting alloys most frequently contain silicon, magnesium, copper, zinc or nickel, alone or in various combinations. Silicon improves the fluidity and castability of molten aluminum; copper and zinc harden the alloy and increase its strength; magnesium improves corrosion resistance, strength, and machinability; and nickel improves dimensional stability and high-temperature strength. The mechanical properties of aluminum casting alloys vary not only with composition but also as a function of casting conditions and subsequent heat treatment, if any. Heat-treated alloys are generally stronger and more ductile than others.

Wrought alloys are divided into two basic classes: non-heat-treatable and heat-treatable alloys. The former rely on the hardening effect of such alloying elements as manganese, silicon, iron, and magnesium for their initial strength. They are further strengthened by various degrees of cold working. Heat-treatable alloys, containing such elements as copper, magnesium, zinc, and silicon, are strengthened by heat treatment and artificial aging, but they may also be cold worked.

Since there are wide variety of aluminum alloys, the Aluminum Association (1984) publishes specifications for wrought and cast alloys and classified them by series, according to the principal alloying elements, as shown in Table B-3.

Table B-3. Designation System for Wrought and Cast Aluminum Alloys

Wrought Aluminum Alloys		Cast Aluminum Alloys	
Alloy Series	Principal Alloying Element	Alloy Series	Principal Alloying Element
1xxx	99.0% Minimum Aluminum	1xx.x	99.0% Minimum Aluminum
2xxx	Copper	2xx.x	Copper
3xxx	Manganese	3xx.x	Silicon plus Copper and/or Magnesium
4xxx	Silicon	4xx.x	Silicon
5xxx	Magnesium	5xx.x	Magnesium
6xxx	Magnesium and Silicon	6xx.x	Unused Series
7xxx	Zinc	7xx.x	Zinc
8xxx	Other Element	8xx.x	Tin
9xxx	Unused Series	9xx.x	Other Element

The application or end product use of the aluminum determines which of these two major alloy categories is employed for the product. Application requirements determine the specific alloying elements and proportions of each element present in the product.

The mix of alloys recovered in aluminum scrap at a given time varies depending on (1) patterns of use and discard of these products, (2) the collection systems that act to intercept the discarded waste materials, (3) the separation efficiency in regard to control of scrap shape and size, and (4) degree of processing required to remove certain contaminants.

New industrial scrap, assuming proper segregation and identification, can be melted with minimal corrective additions. Post consumer scrap, on the other hand, is much more difficult to predict. The nature of aluminum scrap that potentially can be recovered from this scrap stream is variable.

2.4 Secondary Aluminum Industry Structure

Aluminum scrap, in one form or the other, is recovered by almost every segment of the domestic aluminum industry. Independent secondary aluminum smelters, primary producers (integrated aluminum companies), independent fabricators, foundries, and chemical producers can recover aluminum from scrap. As shown in Table B-4, independent secondary smelters and primary producers were the major consumers of aluminum scrap in 1994.

Table B-4. U.S. Consumption of and Recovery from Purchased
New and Old Aluminum Scrap in 1994
(Metric tons)

Class	Scrap Consumption
Independent secondary smelters	1,150,000
Integrated aluminum companies	1,340,000
Independent mill fabricators	728,000
Foundries	103,000
Other consumers	10,900
Total	3,340,000

Source: USBM 1995

2.4.1 Independent Secondary Smelters

The core of the commercial secondary industry is the independent secondary aluminum smelter. The sole purpose of the secondary smelter is to transform aluminum scrap into a marketable product. Independent secondary aluminum smelters consume scrap and primarily produce alloys for the die casting industry and, to a lesser degree, deoxidation products for the steel industry (see Table B-5).

The markets that are served by the secondary smelters are varied but generally fall into the following end use categories (Viland 1990):

Direct Automotive - 22%
Automotive Related - 44%
Small Engine - 8%
Appliance - 7%
Other - 19%.

Automotive uses of aluminum are primarily for transmissions, intake manifolds, heat exchanger systems, wheels, and a variety of smaller parts. Small engine uses include lawn mowers, motor boats, etc.; and appliance uses include parts for lawn mowers and other machinery, hand tools, and small home appliances. The balance of aluminum used in products that are most often seen by the consumer range from cookware to weedwackers.

It is anticipated that the use of aluminum in the automotive industry will grow as automakers seek new ways to save weight and gain fuel efficiency and performance. According to a factsheet, "Aluminum Application in the Automotive Industry," published by the Aluminum Association in October 1993, the use of aluminum by the auto industry has more than double from an average of 35 kg per car in 1971 to 87 kg in 1991. It is estimated that an automobile manufactured in the year 2000 will have at least 136 kg of aluminum (Wrigley 1994).

Table B-5. Production and Shipment of Secondary Aluminum Alloys by Independent Smelters in the United States

(Metric tons)

	1993		1994	
	Production	Net Shipments*	Production	Net Shipments*
Die-cast Alloys:				
13% Si, 360, etc. (0.6% cu, max.)	45,500	44,700	50,500	51,200
380 and variations	518,000	517,000	559,000	560,000
Sand and Permanent Mold:				
95/5 Al-Si, 356, etc. (0.6% Cu, max.)	85,100	84,400	86,400	85,900
No. 319 and variations	67,400	65,700	70,500	71,200
F-132 alloy and variations	24,000	25,800	29,000	29,000
Al-Mg alloys	639	641	639	639
Al-Zn alloys	3,220	3,470	3,530	3,530
Al-Si alloys (0.6% to 2.0% Cu)	10,800	11,000	10,800	10,700
Al-Cu alloys (.15% Si, max.)	1,740	1,730	1,688	1,710
Al-Si-Cu-Ni alloys	1,360	1,400	1,180	1,230
Other	3,790	3,810	2,830	2,860
Wrought alloys: Extrusion billets	80,900	84,900	151,000	152,000
Miscellaneous:				
Steel deoxidation	—	—	—	—
Pure (97.0% Al)	—	—	—	—
Aluminum-base hardeners	93	93	93	93
Other**	34,200	35,200	35,700	35,000
Total	877,000	880,000	1,000,000	1,000,000

* Includes inventory adjustment.

** Includes other diecast alloys and other miscellaneous

2.4.2 Primary Aluminum Producers (Integrated Aluminum Companies)

The other major recyclers of aluminum scrap are the primary aluminum producers (integrated aluminum companies). The types of scrap processed by the primary producers tend to be more segregated than those processed by the secondary smelters. Primary producers participate in either the collection or utilization of new aluminum scrap.

Major primary aluminum producers also operate can recycling programs. These producers have set up thousands of collection centers around the country for used beverage containers. The large-scale aluminum beverage can reclamation programs of these aluminum producers have added substantially to the rate of aluminum recovery from old scrap. The UBC component of old scrap consumption has doubled since 1975 (USBM 1993).

3.0 RECYCLING PROCESSES

Aluminum scrap that reaches the secondary producer is often a mixture of alloys and, therefore, cannot be indiscriminately remelted to make a finished product. Depending on the type of scrap that is received, incoming scrap may be chemically analyzed and or assessed for moisture content and percentage of fines. The scrap is then processed as is required.

Figure B-2 depicts the flow process of a typical smelting operation. The basic steps that include crushing, drying, melting, refining, casting, and reprocessing of dross, are discussed below.

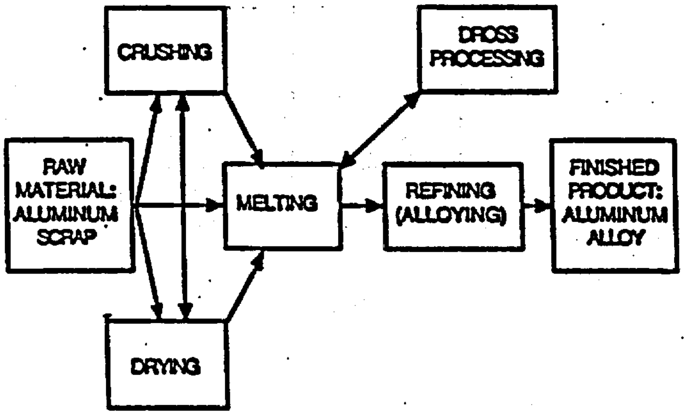


Figure B-2. Smelting Process Flow Diagram
(Source: Viland 1990)

The recycling of used beverage containers is typically conducted by dedicated UBC processing facilities. These processors use only UBC and possibly new scrap to produced can stock ingots, which are refabricated into the same product - beverage containers. Although the basic steps presented in Figure B-2 are also used in a conventional UBC processing plant, additional procedures may be included to accommodate the special handling and equipment necessary to convert nonuniform scrap into more amenable feed. These processors also employ additional equipment to either dry and preheat the scrap or to remove organic materials adhering to the scrap. For example, delacquering systems are necessary in UBC recycling, which typically utilize natural gas and the heat content of the lacquer coating to preheat the scrap and completely pyrolyze the organic coating (Peterson 1995).

3.1 Crushing/Shredding

Crushers and/or shredders are used for reducing the scrap to a more usable size for handling and melting. New scrap, for example, is generated as borings or turnings which are the waste products of machining operations and are often received by the secondary producer as long intertwined pieces that may first require crushing. Typical crushers are also equipped with several sets of magnets at the exit end for the critical removal of iron.

3.2 Dryers

The function of the dryer is to remove contamination, such as cutting oils, plastic, paints, lubricants, etc. This drying process is used primarily to minimize air pollution from the melting furnaces and to reduce the amount of oxidation that occurs while melting. A typical dryer resembles a rotary kiln with an afterburner and baghouse for pollution control. The dryer discharge is often screened to remove fines and then passed over a magnetic separator to remove any iron that may be present.

3.3 Melting Operations

To understand the various methods used for remelting aluminum scrap, it is necessary to characterize the scrap. In addition to the alloy composition (especially magnesium content), two key factors that must be considered in the metal recovery process are the size (surface area) and cleanliness of the scrap.

The configuration of aluminum-base scrap is important because as the surface area of the scrap increases, the amount of oxide on the surface increases. This oxide material is the starting point for a variety of by-products or "slag" (referred to as dross or skim) common to melting operations. The oxide layer increases during the heating and melting process, floats to the surface of the molten aluminum, and is removed. Larger-sized scrap can be melted with lower losses. Finer sizes require greater care and effort to limit losses.

The second important variable in the metal recovery process is the cleanliness of the aluminum-based scrap. Impurities can take many forms: water, dirt, oil, paints and lacquers, sand, tramp metals, rubber, and adhering food and syrups, etc. Impurities are important not only because they represent weight that is not recoverable as metal, but they can often hinder metal recovery.

Figure B-2 presents a two-dimensional field of scrap types based on size and cleanliness. The upper right-hand corner represents the most easily processed scrap - large and clean. This would include cracked ingots, recycled secondary ingot (RSI), and heavy plate scrap. The most difficult to process scraps are identified in the lower left portion of Figure B-2. This category includes decorated foils, used beverage containers, and oily borings and turnings. Generally, as the scrap types move toward the lower left corner of Figure B-2, the cost of processing increases and the metal recovery decreases.

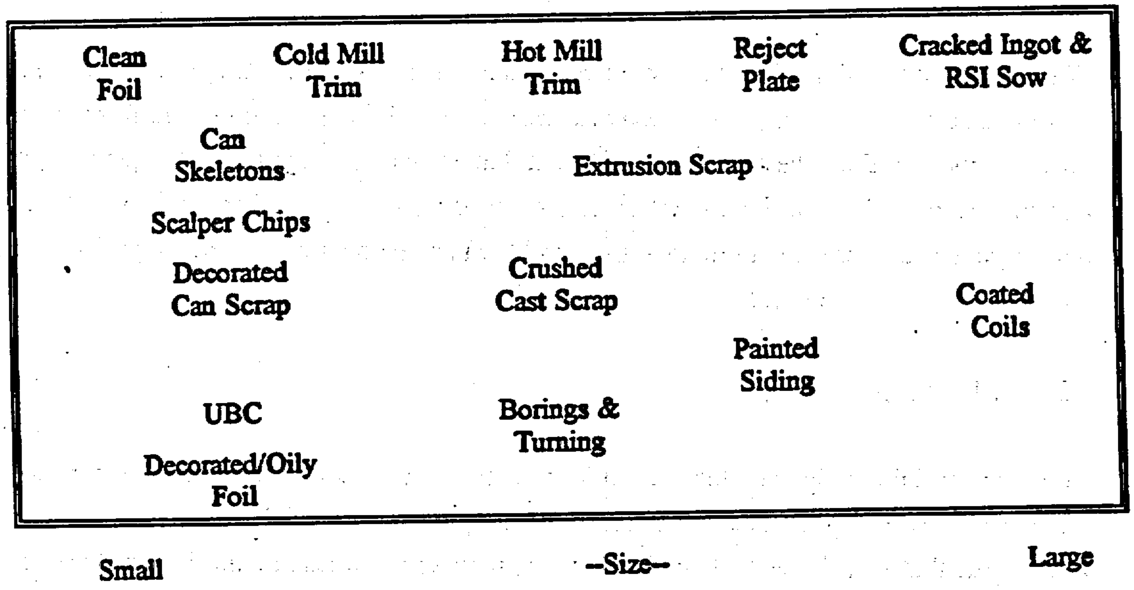


Figure B-2. Aluminum Scrap Types Based on Size and Cleanliness

The characteristics of the scrap to be processed dictate the type of recovery operation used. An induction furnace may be more appropriate for scrap identified in the center and upper left portions of Figure B-2. The dirtier, smaller scraps are typically processed using either a reverberatory furnace with a side-charge well or a rotary barrel furnace. A further discussion of the various furnace types and melting methods used to recovery aluminum from scrap is provided below.

3.3.1 Reverberatory Furnace

The most widely used furnace for melting aluminum-based scrap is the reverberatory furnace. These furnaces are typically natural gas- or oil-fired and can produce up to 9-10 metric tons (mt) per hour with total holding capacity up to 100 mt (Viland 1990).

The rectangular-shaped reverberatory furnaces are commonly used by the secondary aluminum remelters, where dirtier, smaller scrap serves as feedstock. These scraps require melting methods that avoid direct flame impingement or excessively high metal temperatures. Reverberatory furnaces employed by the secondary industry, therefore, consist of two chambers: a larger combustion chamber and an open box-like hearth called a charge well or forewell. The charge well is an open extension of the main or combustion chamber, separated only be a perforated refractory wall, allowing the molten metal level to be the same in each. The charging of the scrap to the furnace is usually done in the charge well. As this chamber is deeper than the combustion chamber, the scrap can be puddled quickly and immersed below the liquid level of the main bath to prevent oxidation. Larger furnaces are disproportionately wider and longer than smaller furnaces because bath depth is about 30 inches, regardless of furnace size (Neff 1991; Lauber, et al 1973).

Roof height above the molten metal depends on the height of the charging door, which is dictated by the kind of charge used. Roof height also depends on the heat-release factor relating the furnace volume to the heat input. In general, furnace builders prefer no more than 30,000 Btu/ft^3 of space above the bath (Planson 1995).

The heating source in the reverberatory furnace is located directly above the metal. Most reverberatory furnaces use a nozzle-mix burner that throws a long flame, making use of "double-pass firing." It begins with a luminous or semiluminous flame, relatively high in the combustion chamber, that radiates heat to the refractory walls and roof. As the walls and roof become incandescent, they radiate heat to the bath. On the return path to the flue, in the same wall as the burners, convective heat transfers from the gases. This radiation on the outgoing path and convection on the return path provide a double transfer of heat (Planson 1995).

The reverberatory furnace with a side-charge well has found wide acceptance among the secondary industry. In this furnace, a large pool of metal is heated in the hearth by gas-fired burners. The hot metal travels to the side-charge well where it meets the scrap. It loses its heat in the melting of the scrap and returns along with the newly melted metal back to the hearth for reheating.

Melting of aluminum scrap materials is greatly facilitated by providing forced convection within the melt using a bath circulation system. Molten metal circulation pumps are commonly employed to improve productivity, energy efficiency, alloying and temperature homogeneity, and lower melt loss (Neff 1993).

Although there are numerous variations in methods used by different operators, smelting of aluminum scrap in reverberatory furnaces usually includes the following seven steps: (1) charging scrap to the furnace, (2) blending and mixing with alloying materials, (3) addition of fluxing salts, (4) demagging or removal of magnesium, (5) degassing, (6) skimming, and (7) pouring or tapping (Garino 1987).

Charging. A heat or cycle begins with the charging of scrap to the furnace, depending on the melt rate desired, type of scrap being processed, and other factors. Temperatures in the furnace vary from 750°C to 1200°C in the combustion chamber and from 650°C to 800°C in the charge well.

Most operators leave about 20 to 40 percent of the molten metal from the previous heat in the furnace to aid in quickly melting the next charge. This molten material, called the "heel," shortens the cycle by several hours. The furnace is fully tapped, however, when different melts or metal compositions are being poured. Most large, modern smelters avoid this by continually producing the same metal compositions from a particular furnace from heat to heat and a heel is almost always left in the furnace.

Scrap is added to the charge well of the furnace, either in batches with a front-end loader or continuously from a conveyor belt feeder. Feeding method is dictated by the type and form of the scrap being charged. Charging in batches is usually necessary for mixed scraps. The heavy scrap is charged and melted first, followed by the finer material. Charging time usually required 16 to 18 hours. It is dependent upon furnace size, scrap type, and other circumstances. Demagging and degassing processes, which will be discussed in further detail below, usually require from 2 to 4 hours, and tapping, if a holding furnace is not used, typically requires 3 to 4 hours. The total cycle requires about 24 hours (Garino 1987).

Blending. The production of secondary aluminum alloys is essentially a process of blending various scrap aluminum alloys until the proper specifications are achieved. As the furnace nears capacity, scrap alloyed with metals needed to bring the melt or batch to specification, such as copper, silicon, manganese, magnesium, zinc, etc., or the alloy metal alone are added and blended into the melt. Mixing scrap to meet desired specifications has limitations. Magnesium is the only metal commonly alloyed with aluminum that can be economically removed in the secondary smelter, if in excess. Other commonly alloyed metals such as iron, silicon, copper, zinc, and manganese cannot be economically removed. To control alloy content when metals other than magnesium are in excess of specification, the melt must be diluted with pure aluminum scrap. Electrical wire or cable is commonly used for this purpose.

Fluxing. Flux is a substance used to free metals from oxide, promote their coalescence, and act as a protective coating for certain molten metal baths. Fluxing salts most commonly used include a mixture of sodium chloride (NaCl), potassium chloride (KCl), and a fluoride salt, usually cryolite (Na_2AlF_6). The mixture is usually about 48 percent NaCl, 48 percent KCl, and 4 percent fluoride salt.

The primary purpose of a flux in the smelting furnace is to cover the molten aluminum metal which prevent oxidation and hydrogen gas absorption. In addition to preventing oxidation and gas absorption, the flux also absorbs most of the various contaminants contained in the scrap and formed during the smelting process (i.e., residues of burned paint and coatings - mostly titanium oxides, ash, dirt, and other nonmetallics). Failure to remove these contaminants impairs the mechanical properties of the metal, especially its castability.

Once contaminant materials are absorbed or entrapped in the flux, they form a solid crust that can be removed by skimming from the surface of the molten metal. This is usually done with a perforated ladle or long handled cup-like device with holes in it to allow any molten metal to drain out.

The amount of flux added varies with the about of contamination in the scrap. Usually about 1 kg of flux is added for each kilogram of nonmetallics estimated to be in the scrap. For scrap estimated to contain about 6 percent contaminants, about 60 kg of flux would be added per metric ton of scrap processed (Crepeau 1992).

Effluents. Air emissions from reverberatory furnaces are usually segregated into separate streams. Emissions from the combustion chamber of the furnace consist of products of combustion and are vented directly to the atmosphere. Air emissions from the charging well are typically vented to baghouses, which are often coated with lime or a similar commercial product that control both fumes and particulates.

3.3.2 Rotary Furnace

Low grade aluminum scrap and light scrap are best processed by rotary furnaces in which the charge materials can be melted and refined, aided by the rotating action of the furnace. U.S. companies often employ small rotary furnaces for production of recycled secondary ingot sows that are sold either to large producers of aluminum foundry alloys or directly to the producers of castings. The aluminum produced from the melting of these low grade materials is cast in sows for further applications.

The basic principle of the rotary furnace is to melt the salt flux and to coat the aluminum metal particles with the flux to avoid oxidation. The rotation of the furnace accomplishes this by forcing the dross beneath the surface of the bath and away from the direct burner flame as quickly as possible. The furnace burners, which can be fired with natural gas or fuel oil, are used to heat the exposed refractory, which exchanges heat to the bath during the rotation cycle.

A rotary furnace typically consists of a refractory-lined barrel in which there is a burner in one end that normally coincides with the charging end and at the other end a flue where the fumes are exhausted. Smaller furnaces may have the burner and flue on the same end. Off-gases are normally cleaned in baghouses.

Melting Process. In a conventional rotary furnace, the aluminum scrap is charged into the furnace with a fluxing agent, usually a mixture of sodium chloride and potassium chloride. The burners are set at high (reaching temperatures up to 850°C) and the barrel is rotated to ensure intimate mixing of the flux and metal. As the charge is heated, the aluminum melts prior to the flux and remains at the bottom of the furnace with very little movement. The flux floats on the molten metal preventing oxidation. Once liquid, the flux disperses and fluidizes the oxides, facilitating their extraction from the furnace (Artola and Roth 1990).

After the materials have been melted, rotation of the furnace is stopped and the molten aluminum is withdrawn from a tap hole in an end of the furnace. After withdrawal of the molten aluminum, the salt is similarly withdrawn from the end of the furnace by gravity.

A hydraulically-actuated tilt mechanism is often employed to tilt the furnace in order to facilitate the removal of the molten aluminum and salt from the furnace. As the furnace has substantial weight, up to 60,000 pounds, and diameter, generally 8 to 10 feet, the tilt mechanism is complex and adds considerable cost to the unit.

Other commonly used furnaces have an outer cylindrical drum and a tapered refractory lining. By use of the tapered refractory lining, the molten materials can be drained from the furnace without the necessity of a tilt mechanism (Evans 1982).

Design Characteristics. Typical rotary furnaces range in size from 4 to 55 mt of capacity of aluminum and flux. Table B-6 shows the cycle characteristics of rotary furnaces of varying sizes .

Table B-6. Typical Cycle Characteristics of Various Rotary Furnaces*

Furnace Size	Cycle Time (Hrs)	Cycles per Day	# Al/Hr Produced	Al lbs. Prod./Day	BTU/lb** Charge	BTU/lb** Al
4 mt	2.25	10.6	1,550	37,200	1,150	2,880
15 mt	4.25	5.6	3,550	85,200	900	1,900
40 mt	6.25	3.8	6,400	153,600	800	1,790
55 mt	7.50	3.2	7,400	177,600	780	1,700

* Assumed 60 percent concentrate of Al in charge.
** The BTU/lb values are based on metered fuel use.
Source: Artola and Roth, 1990.

Salt Cake. The drawback of a rotary furnace is the higher requirement for fluxing salt usage and the subsequent generation and disposal of residue, known as salt cake. In a conventional rotary salt furnace using a charge of dross containing about 55.5 percent metal and 45.5 percent oxides and weighing 12,000 pounds, the weight of the salt flux charge would be 6,000 pounds. At the end of the melt cycle, the furnace yields 6,000 pounds (50% of the total dross weight) of metal assuming a 90 percent recovery efficiency. The remaining quantity, minus melting losses, represents salt cake (Johnson 1990).

Plasma Arc Treatment Process. The secondary aluminum industry is under increased pressure to reduce or eliminate the land disposal of its salt slags generated from rotary furnace processing of dross and scrap aluminum. The plasma arc treatment process is a new technology for recycling aluminum dross and scrap that eliminates the use of flux in a rotary furnace fitted with a plasma arc torch. This process will avoid the generation of salt cake and make the residue more suitable for low cost landfilling.

The process is based on utilization of a plasma gas arc heater. Inside the plasma torch arc are two tubular electrodes placed end-to-end but separated by a small gap. During the operation, a process gas (i.e., air, argon, or nitrogen) is injected into the small gap between the electrodes. When high voltage power is applied, an electric arc is initiated between the electrodes. This arc heats the incoming gas at temperatures in excess of 5000°C. At this temperature, the gas is dissociated and partially ionized. This ionized gas is called a plasma. The plasma is ejected out of the torch inside the furnace and heats the dross. Since there is no combustion involved, virtually any gas can be used. Because of the extreme temperatures reached, the same energy can be transferred using 10 to 20 times less gas than a fossil fuel burner (Lavoie, et al, 1990).

3.3.3 Induction Furnace

The induction furnace melts metal by means of an electric current which ensures uniform composition of the melt due to magnetic stirring by induced currents. Although electric induction furnaces are used by some secondary aluminum plants, their use is limited for a number of basic reasons:

- These furnaces are small in size (less than 5 mt) and not well suited for the reclamation of low grade aluminum materials where salt fluxes are required.

- Due to the high heat requirement of melting aluminum scrap (approximately 600 kW/t), the melting cost of the induction furnace is much higher as compared to fuel costs in the reverberatory furnaces.

- A high power rating is required for the induction furnace to match the speed of melting of a reverberatory furnace.

- A much higher investment cost is required for an induction furnace installation to equal the output of the reverberatory furnace.

When induction furnaces are installed in the aluminum reclamation industry, it is usually for melting high grade (new) scrap materials or in instances of secondary aluminum ingot manufacturing of certain alloys where the electro-magnetic stirring action can be used to advantage (Roscrow 1983).

3.4 Refining Processes

3.4.1 Demagging

In the secondary aluminum smelting industry, the availability of old scrap often includes a predominance of wrought scrap products whose overall magnesium content exceeds that required for producing specification die cast or foundry alloy ingot. Correspondingly, the melt must often be "demagged." Magnesium is removed from the alloy melt almost exclusively by a chemical displacement reaction using chlorine gas. Reverberatory furnaces generally use a gas injection pump to introduce chlorine gas to the melt. The elemental chlorine gas reacts selectively with the magnesium forming magnesium chloride ($MgCl_2$), which at normal furnace temperatures is a liquid that rises to the surface and is adsorbed by the flux. In such chemical displacement reactions, only metals above aluminum in the electromotive series can be separated and removed from the melt because of their greater reactivity or affinity for the anion of the formed salt, in this case, chlorine. Metals below aluminum in the series are less reactive and, therefore, cannot be separated by displacement reactions as the chlorine will react or combine with aluminum first (Neff 1993).

During the demagging process aluminum chloride ($AlCl_3$) is also formed, especially if the chlorine added exceeds the stoichiometric amount required to form magnesium chloride or if mixing conditions are imperfect. $AlCl_3$ is extremely hygroscopic and reacts with moisture in the air to form hydrogen chloride (HCl) or hydrochloric acid. Some chlorine gas (Cl_2) also escapes. The occurrence of the compounds in the fumes given off from the demagging process require treatment, usually with caustic wet scrubbers, before release to the atmosphere.

3.4.2 Degassing

Aluminum, like most molten metals, absorbs hydrogen gas from moisture in the air, which if not removed separates out during solidification causing blisters or other imperfections in the metal. Molten aluminum absorbs hydrogen in proportion to its temperature. Therefore, degassing is usually the last process completed before skimming and casting, after the melt temperature has stabilized slightly above its freezing point.

In addition to removing magnesium from the melt, as previously described, chlorine is also very effective in removing gaseous contaminants, such as hydrogen, as well as oxides, ash, dirt, and other nonmetallic inclusions. These materials are carried to the surface where they escape to the vent system or are entrapped or adsorbed in the flux and are skimmed off with the flux (Crepeau, et al, 1992).

Other degassing measures can be used as well. Hydrogen may be removed from the melt in the furnace using hexachloroethane tablets, degassing fluxes, or other in-line degassing/filtration systems positioned between furnace and casting pits (Neff 1993).

3.5 Casting

The aluminum produced by the secondary industry is principally cast into ingots or sows. These ingots are then shipped to fabricating plants where the final product is produced using one of three casting method: die casting, permanent mold casting, and sand casting. Based on production values shown in Table B-5, secondary aluminum smelters produced approximately 61 percent diecast alloys and 21 percent sand and permanent mold alloys in 1994. A brief definition of these three casting methods is provided below:

- Die casting forces molten metal into a steel die or mold, under pressure and is normally used for high-volume production. Accurate parts, requiring a minimum of machining, can be reproduced.

- Permanent mold castings involves molds and cores of steel or other metal. In this process, molten aluminum generally flows into the mold by gravity, although vacuum is sometimes applied. Permanent mold castings can be made stronger than either die or sand castings.

- Sand casting is the most versatile method. Virtually any pattern can be pressed into a fine sand mixture to form a mold into which the aluminum is poured. It is a slower process, but usually more economical for small quantities, intricate designs, or when a very large cast is needed.

3.6 Reprocessing of Dross

Dross is a byproduct of all melting processes and refers to the "scum" that forms on the surface of molten metal largely because of oxidation and sometimes by impurities rising to the melt surface. The quantity of dross produced ranges from 1 to 2 percent in holding furnaces of primary smelters and to 3-4 percent in scrap melting furnaces. In most cases, dross skimmed from the furnace will contain large quantities of metallic aluminum which can reach up to 80 percent of the weight of the material skimmed (Lavoie, S., et al, 1990).

3.6.1 Types of Dross

Fluxing techniques and production practices vary widely throughout the industry, resulting in the production of different types of dross. The drosses produced from the various melting/remelting methods have been described in the literature (Aluminum Association 1994; Kulik and Dale 1990) and are summarized below.

White Dross. The generation of white dross occurs at primary aluminum smelters, extruding plants, sheet mills, foundries, and die casters. The furnaces at these facilities are operated without fluxing, and the dross skimmed from the furnaces is grey or metallic white in color. White dross, as it is skimmed, has a very high aluminum metal content. However, oxidation or "thermiting" will occur very rapidly because of the high temperature and absence of flux. If not controlled, all of the aluminum metal could convert to aluminum oxide. Thus, thermiting has a major impact on the value of the metal recovered from white dross and the amount of waste aluminum oxide generated. Depending on the efforts made to control thermiting at the producers' plant, recovery of metal from white dross can be as high as 80 percent or as low as 15 percent.

Black Dross. Black dross is generated by secondary aluminum smelters who use open hearth reverberatory furnaces for melting old castings, clippings, turnings, and UBCs. A salt/potash flux is used in the open hearth to reduce the amount of oxidation occurring on the exposed metal. At the high molten metal temperatures, the flux melts and becomes dark colored, hence the name "black" dross. The diluting effect of the added flux results in a metal recovery of 12 to 18 percent.

Salt Cake. Recovery of aluminum metal from white and black dross is accomplished in molten salt bath rotary furnaces. A salt-potash flux is used in the furnace to maximize the recovery of the metal, and the spent flux is discharged from the furnace either continuously or on a batch basis. This residue, containing 3 to 5 percent metal, is referred to as salt cake. Salt cake may be economically reprocessed for aluminum metal recovery if the dross processor has the necessary crushing and concentrating equipment. If not reprocessed, the salt cake is disposed of in landfills.

The flow diagram shown in Figure B-3 identifies the various by-products that result from the melting or handling of aluminum.

3.6.2 Recovery of Aluminum from Dross

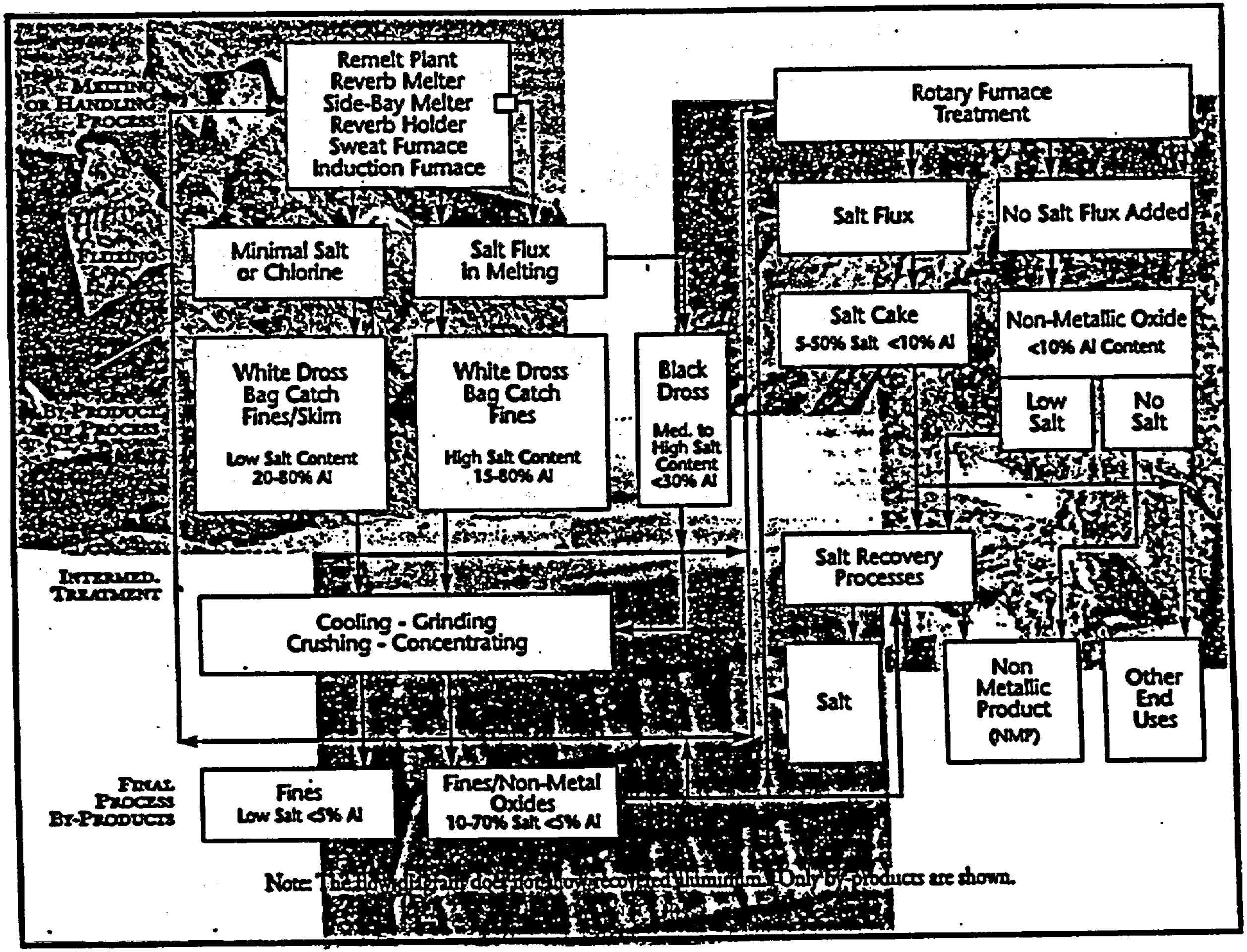
As previously discussed, the rotary furnace is used for recovery of aluminum metal from dross. Some high-grade metallic slabs and pieces may be hand-sorted from the dross and fed to reverberatory furnaces, but the greatest tonnage of dross is charged into rotary furnace.

Most large dross processors have installed concentrating facilities to upgrade black drosses to a 60 to 70 percent concentrate before being charged to the rotatory furnaces. In most situations, metal cannot be economically recovered from black dross without concentrating. The concentration process involve the following operations:

- Crushing. Black dross is skimmed from reverberatory furnaces into boxes, tubs, and containers of a variety of shapes. The dross freezes into these shapes and must be crushed into pieces no larger than 5 to 8 inches in size. Jaw crushers or impactors are used for this purpose.

- Milling. Crushed dross is milled to free aluminum metal from aluminum oxides and salt particles. This can be accomplished in a hammer mill, ball mill, or autogenous mill where pieces of large aluminum metal act as the grinding media. Most milling is done dry so baghouses are used to control dust generation. In some situations, wet milling is acceptable and this can result in a higher grade concentrate because the water dissolves the flux clinging to the aluminum particles. Wet milling can produce a brine suitable for subsequent flux recovery and low-chloride mill products suitable for marketing.

- Screening. The mill discharge is passed over vibrating or trommel screens to recover the aluminum concentrate and other potentially marketable products. For example, the +12 mesh material will be concentrate, and the -30 mesh fraction the tailings. The intermediate product may contain 15 to 25 percent aluminum metal and is valuable as an exothermic product. Tailings from wet milling are low in chlorides and suitable for cement manufacturing (Kulik and Dale 1990).



Note: The flowdiagram does not show recovered aluminum. Only by-products are shown.

Figure B-3. Aluminum Recovery By-Products

Figure B-4 is a schematic flowsheet showing a typical dross processing operation.

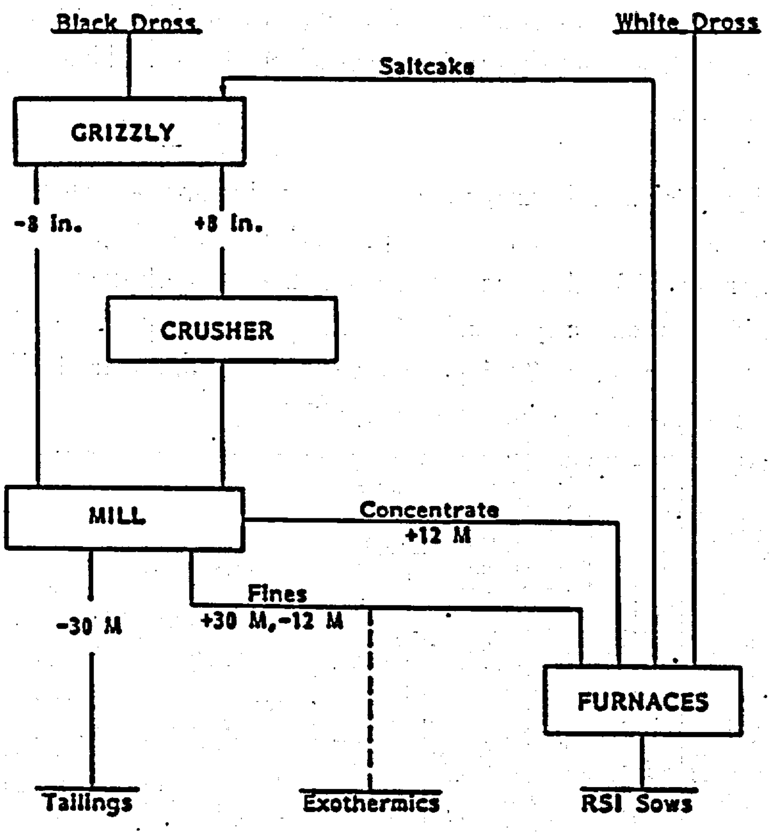


Figure B-4. Dross Processing Schematic Flowsheet

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APPENDIX C

RECYCLING OF COPPER SCRAP

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EXECUTIVE SUMMARY

Copper ranks third in world metal consumption after steel and aluminum. The major copper consuming nations or areas of the world are Western Europe (29%), United States (20%), Japan (15%), the C.I.S. (7%), and China (6%). About two-thirds of the metal consumed in the United States was for construction and electrical/electronic uses, finding widespread application in all sectors of the economy. Machinery, transportation, and other miscellaneous end use sectors make up the remainder: copper and copper alloy powders are used for brake linings and bands, bushings, instruments, and filters in the automotive and aerospace industries; for antifouling paints and coatings; and for various chemical and medical purposes. Copper chemicals, principally copper sulfate and the cupric and cuprous oxides, are widely used as algacides, fungicides, wood preservatives, copper plating, pigments, electronic applications, and numerous special applications.

In 1993, about 555,000 metric tons (mt) of copper-base scrap was consumed in the United States and made up about 22% of U.S. apparent copper consumption. Most U.S. scrap is consumed at brass mills, smelters, and into makers. Wire rod mills, by contrast, consume 77% of the U.S. refined copper, but consume very little direct melt scrap.

Sources of Secondary Copper. The Institute of Scrap Recycling Industries, Inc. (ISRI) and the National Association of Recycling Industries (NARI) recognize various classes of copper and copper alloy scrap. The major unalloyed scrap categories are No. 1 copper, which contains greater than 99% copper and often is simply remelted, and No. 2 copper, which usually must be re-refined. In addition to the many copper and copper alloy scrap types, there are many special types, such as skimmings, ashes, and residues, which contain 12% to 30% copper; and others of lower copper content, such as electronic scrap, refining slags, printed circuit and other clad materials, and metal-laden waste liquors.

The availability of secondary copper is linked with the quantity of products consumed and their life cycles: copper in electrical plants and machinery averages 30 years; in nonelectrical machinery, 15 years; in housing, 35 years; and in transportation, 10 years. The average useful life for copper products is about 25 years, before being scrapped and entered into the market as old scrap. The rate of old scrap recovery is limited not only by copper's long life and its essential uses, but also by the sensitivity of scrap collection to market prices.

Recovery Methods. Most old scrap must be reprocessed by either smelting and refining to form a pure copper product. Fire refining in reverberatory or other furnace may be sufficient for the better grades of scrap. The fire-refining process uses oxidation, fluxing, and reduction to produce refined ingot, wire bar, slab, or billet. For higher grades of refined copper cathode, however, the poorer grades of scrap must be first smelted with various fluxes and cast into anode form for further processing in an electrolytic refinery. Byproducts, such as tin and precious metals, may be retrieved during the preliminary procedures of smelting, or during refining from the tankhouse sludges. Other impurities, such as iron, lead, arsenic, and antimony may be removed in the slag by fluxing. Reverberatory or electric rotary melting furnaces are used for casting various copper forms, such as slabs, cakes, or billets.

Black copper (75% to 80% copper) is the principal product of the blast furnace and still contains some iron and zinc along with most of the tin, lead, nickel of the charge. Traditionally, this material is refined in a scrap converter, which is of a more modest size than its primary cousin; also coke is added liberally to the charge, adding extra heat and providing a mildly reducing condition, thus facilitating removal of zinc, tin, and lead in the gas stream. A copper anode is poured for final refining in electrolytic tankhouse.

The final stage in copper purification involves an electrolytic process in which copper anodes are suspended in acid along cathode starting sheets. By passing a direct electrical current between copper anodes and corresponding cathode starter sheets, copper dissolved from the anode is plated out in purified form on the cathode. Electrorefining yields copper with less than 40 ppm impurities.

RECYCLING OF COPPER SCRAP

1.0 SALIENT COPPER STATISTICS

In order to assess future potential impacts of recycling copper scrap from nuclear facilities, it is important to understand the recycling process but also the quantitative and dynamic relationship between not only production of copper from mined ore and from the recycling of copper and copper alloys. The most current and comprehensive data regarding copper production and recycling are those compiled by the United States Bureau of Mines (USBM) as reported in annual reports.

Table C-1 provides a summary of salient copper statistics for the United States for the years 1989 through 1993 (USBM 1995a). Copper is currently mined in 55 countries of which the top two, Chile and the United States, account for more than 40% of primary (new) copper production (Bowlby 1994). For 1993, domestic mines yielded 1.8 million metric tons or 19% of the worlds mined copper production. The principal mining states in descending order, Arizona, Utah, New Mexico, Michigan, and Montana, accounted for 98% of domestic mined copper production. While U.S. copper was recovered at 50 mines, 15 mines accounted for about 95% of domestic production. Total U.S. mine capacity in 1993 was estimated at 2.06 million tons.

Most copper mine-producing countries, including the United States, engage in export and import of copper at various stages of copper processing and refining, which extend from copper ores (<1% copper) and copper concentrates (containing between 18% and 40% copper) to highly refined copper (>99%). Copper ores and concentrates are exported to other countries for further processing and refining for their domestic use or may again be exported. Table C-1 also reveals that secondary copper production from recovered old scrap in recent years accounts for about 22% of total copper consumption in the United States.

Table C-1. Salient Copper Statistics for the United States

	Quantities (Metric Tons x 10^3)				
	1989	1990	1991	1992	1993
Primary (new) Copper from Domestic Ore	1498	1588	1631	1765	1801
(% of world total)	(16)	(17)	(18)	(19)	(19)
Secondary Copper Recovered from Old Scrap	547	537	518	555	555
Copper Exports:					
Refined	130	211	263	177	216
All Other	725	780	806	676	685
Copper Imports:					
Refined	300	262	289	289	343
All Other	515	512	512	593	637
Stock Inventories:					
Refined	107	101	132	205	153
Other	132	119	135	166	145
Annual U.S. Copper Consumption*	2203	2182	2144	2359	2535

* Consumption = primary refined production + copper from old scrap + refined exports - refined exports ± changes in refined stock inventory.

Source: USBM 1995a

The most important properties of copper that account for its diverse utilization include electrical and thermal conductivity, strength with good formability, corrosion and stress-corrosion resistance, ease of joining, color, recyclability, and cost. A significant percentage of copper issued in the making of metal alloys have a wide range of copper content. It should be noted, however, that the conductivity, stress-corrosion resistance, ductibility, and recycling capability of copper are not improved by alloying. Hence, the aim of alloying is to change or improve one of the other properties with minimum degradation of the desirable properties of pure copper. For 1993, U.S. consumption of copper by end-use sections is defined in Table C-2. Construction was the dominant end use and is heavily influenced by economic factors that dictate housing starts, which in 1993 rose by 7.1% over the previous year and accounted for 42% of domestic copper consumption. Electric and electronic products remained unchanged and accounted for 24%.

Table C-2. Relative Consumption of Copper Products by End-Use Sector for 1993

End-Use Sector	Relative Consumption (%)
Building Construction (electrical, piping, structural)	42
Electrical Products	24
Industrial Machinery and Equipment	13
Transportation Equipment	12
Consumer and Miscellaneous	9
Total	100

Source: USBM 1995a

2.0 CLASSIFICATION AND QUANTITIES OF COPPER SCRAP

The Institute of Scrap Recycling Industries (ISRI) and the National Association of Recycling Industries (NARI) recognize various major classes of copper scrap (NARI 1980; Newell 1982; Riley 1983). The major unalloyed scrap is termed No. 1 copper, which must contain greater than 99% copper and No. 2 copper, which must contain a minimum of 94% copper. For copper alloys, ISRI has identified 50 separate classifications. Additional classifications exist for copper containing waste streams, such as skimmings, ashes, and residues generated in copper smelting and refining processes.

Copper scrap is further categorized as either "old" or "new" scrap. New scrap or manufacturing scrap is generated during the fabrication of copper products. For example, copper containing end-products manufactured from semifabricates, such as copper sheets, strips, piping, or rod, may have product yields that in some cases are as low as 40%. Thus, new scrap materials generated in the form of borings, turnings, stampings, cuttings, and "off-specification" products are commonly sold back to the mills producing the original semifabricate from which the new scrap was generated. For obvious reasons, new scrap or manufacturing scrap is not considered a new source of copper supply. The need to discuss new scrap, however, is dictated by the fact the some quantitative data reported by scrap processors do not distinguish between new and old scrap.

"Old scrap" is generated from worn-out, discarded, or obsolete copper products and, thus, is considered to be a new source of supply. Since World War II, the ever increasing reservoir of copper products in use has increased drastically both in the U.S. and globally. The U.S. scrap reservoir of items in use or abandoned has increased from 16.2 million tons in 1940 to nearly 70 million tons in 1991 (USBM 1993a). World wide, the reservoir of copper materials in use or abandoned increased from 32.9 million tons to about 190 million tons during this period (Figure C-1).

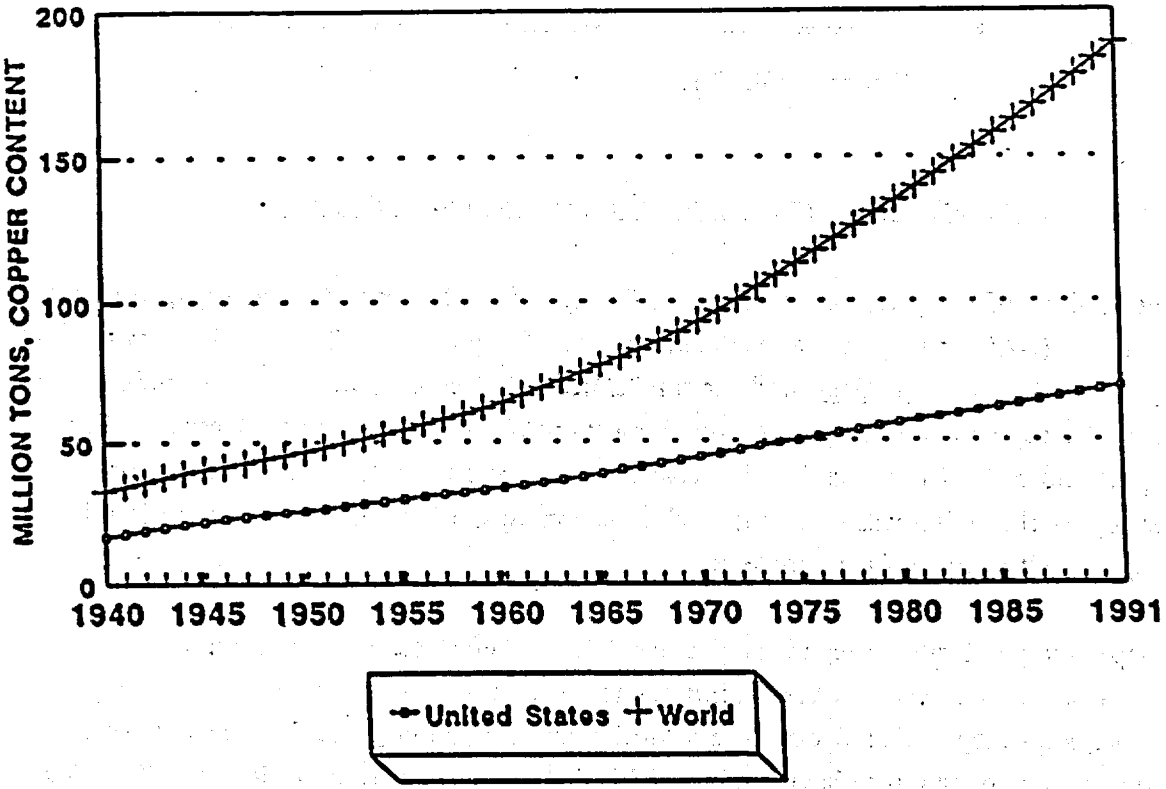

Figure C-1. U.S. and World Scrap Resource Pool of Copper Materials in Use
Source: USBM 1993a

Copper has few application that are dissipative by nature; it is estimated that only about 0.5% of total copper consumed may be unretrievable, as in the case for copper-containing chemicals. Estimates of the recycling efficiency of this copper reservoir, however, are complicated. The availability of copper scrap is linked with the quantity of copper-containing products and their life-cycles. Estimates of life-cycles have been made in behalf of major products: copper in electrical plants and machinery averages about 30 years; in non-electrical machinery, 15 years; in housing, 40 or more years; and in transportation, 10 years (USBM 1993b; USBM 1995b; Glockman 1992).

In 1992, the U.S. Bureau of Mines estimated that the recycling efficiency for copper was about 30%. Recycling efficiency was defined as the quantity of old scrap recycled divided by the quantity of old scrap theoretically available, using average product sector life-cycles and adjusted for net scrap trade. The rate of old scrap recovery is limited not only by copper's long life-cycles and its essential uses but also by economic factors related to recovery costs and market prices. Consequently, recovery of some products, such as buried cables, may be delayed until such time as market prices permit the cost and time needed to recover them.

New scrap, on the other hand, has a relatively short life of about 30 days, and its recovery is limited by domestic manufacturing rates and supplies of inventories.

Table C-3 provides summary data regarding total quantities of new and old copper scrap recycled in the United Stated between 1989 and 1993 as well as their market value for the corresponding years. These quantities are defined by the copper content of scrap material without defining the composition of the scrap. A breakdown of the type of scrap recycled is provided in Table C-4. For both new and old scrap, the overwhelming quantity is contributed by copper-base scrap in which copper is either the exclusive or dominant metal constituent. Copper-base alloys are principally brass and bronze, which on average contain about 80% copper and varying percentage of zinc, tin, lead, nickel, and aluminum. Collectively, new and old copper-base scrap recycled in 1993 contributed about 1.2 million metric tons.

Table C-3. Quantities of Recycled Copper in the U.S. and Their Economic Value

Year	Quantity in Metric Tons (% Total)			Total Value (Thousand Dollars)		
	New Scrap	Old Scrap	Total	New Scrap	Old Scrap	Value per Unit Weight ($/kg)
1989	761,000 (58)	548,000 (42)	1,308,000	2,198,000	1,581,000	2.88
1990	774,000 (59)	537,000 (41)	1,311,000	2,101,000	1,457,000	2.71
1991	682,000 (57)	518,000 (43)	1,201,000	1,645,000	1,250,000	2.41
1992	723,000 (56)	555,000 (44)	1,278,000	1,712,000	1,313,000	2.37
1993	731,000 (57)	555,000 (43)	1,286,000	1,475,000	1,120,000	2.02

Source: USBM 1995b

Table C-4. Scrap Processed in the U.S. by Type of Scrap (Metric Tons)

Type of Scrap	1989	1990	1991	1992	1993
New Scrap:					
• Copper-base	737,088	750,707	660,550	697,471	702,360
• Aluminum-base	23,761	23,092	23,092	25,242	28,403
• Nickel-base	45	42	42	72	117
• Zinc-base	—	—	—	—	—
Total	760,894	773,841	682,289	722,785	730,880
Old Scrap:					
• Copper-base	530,499	502,326	495,397	523,172	521,434
• Aluminum-base	16,957	34,303	22,921	31,372	33,323
• Nickel-base	78	77	61	46	41
• Zinc-base	27	26	22	18	17
Total	547,561	536,732	554,608	554,608	554,815
Grand Total	1,308,455	1,310,573	1,200,690	1,277,393	1,285,695

Source: USBM 1995b

A significant effort in the recovery of copper scrap is the segregation of scrap on the basis of composition prior to reprocessing. For economic reasons, unalloyed or refined copper scrap is commonly reprocessed separately from alloyed copper. (The various reprocessing methods applicable to unalloyed and alloyed copper scrap are discussed in detail in Section 3.0 below.) Table C-5 identifies the quantities of scrap recovered in terms of the form. Consistently, the largest percentage of recovered copped comes from copper-base scrap in the form of brass and bronze alloys (55-60%) and as unalloyed (refined) copper (~37%).

For 1993, reprocessing of copper scrap involved (1) nine electrolytic refineries and six fire refineries; (2) 28 ingot makers of brass and bronze ingots, (3) 35 brass and wire-rod mills, and (4) about 160 foundries, chemical plants, and miscellaneous manufacturers (USBM 1995b). Copper refiners recovered about 38%; brass and wire-rod mills about 48%; brass and bronze ingot makers about 10%; and foundries, miscellaneous manufactures, and chemical plants less than 5% (Table C-6).

Table C-5. Form of Recovery of Recycled New and Old Copper Scrap

Recovered Form of Copper	Quantities (Metric Tons)				
	1989	1990	1991	1992	1993
As Unalloyed Copper	489,282	449,901	426,087	442,503	469,601
In Brass/Bronze Alloys	774,770	800,711	727,618	776,981	753,968
In Other Alloys	42,455	57,181	44,964	57,617	61,909
In Chemical Compounds	1,948	2,720	2,021	292	217
Total	1,308,455	1,310,573	1,200,690	1,277,393	1,285,695

Source: USBM 1995

Table C-6. Copper Recovered as Refined Copper and in Alloys and Other Forms from Copper-Base Scrap Processed in the U.S., By Type of Operation for 1993

Type of Operation	From New Scrap	From Old Scrap	Total
Ingot Makers	34,708	92,102	126,810
Refineries	112,707	347,081	459,788
Brass & Wire-Rod Mills	534,786	50,701	585,488
Foundries & Manufacturers	19,941	31,602	51,543
Chemical Plants	217	—	217
Total	702,360	521,486	1,223,846

Source: USBM 1995b

3.0 COPPER PRODUCTION

Copper and various copper alloys are produced from three major raw materials: (1) sulfide copper minerals, (2) oxidized copper minerals, and (3) scrap metals containing varying amounts of copper. These materials are processed pyrometallurgically and/or hydrometallurgically to produce a high-purity (> 99.9% Cu) electrorefined copper suitable for all electrical, electronic, and select mechanical uses. Some lower purity or "fire-refined" copper is produced for lesser demanding copper uses inclusive of copper alloys. In some cases, copper alloy scrap may be directly transformed into new alloys similar to the original scrap composition . The production of refined copper from primary ores and from secondary copper materials share common processes in all but the initial process of concentrating ore (i.e., beneficiation). This section provides an overview of the primary and secondary processes used in the production of refined copper and other copper products.

3.1 Beneficiation and Copper Ores

Copper minerals exist in relatively low concentrations as copper sulfides or oxides. Worldwide, the average copper contents of ores range from less than 0.5% Cu (open pit mines) to slightly greater than 3% Cu (underground mines). Copper ores currently mined in the United States are largely sulfide bearing and typically contain about 0.5% copper. Ores this dilute in copper cannot be smelted without prior concentration. Thus, the first step in copper production from ore is physical beneficiation of the ore to produce a copper concentrate (Davenport 1986).

Copper Oxide Ores. Less than 10% of primary copper originates from oxidized copper ores, principally oxides, silicates, sulfates and carbonates. It is virtually all produced hydrometallurgically by leaching the ore in aqueous H_2SO_4 followed by electrodeposition or cementation. The leaching is carried out on heaps of ore, in vats, or occasionally in agitated leach tanks. The chemical process is the same for all the leach methods - the minerals are dissolved go Cu^{24} ions in a Cu_4-H_2SO_4-H_2O solution.

The solutions form the leaching processes contain 0.5-50 kg m^{-3} copper. This copper is recovered by either cementation on steel scrap or, as is more often the case, by preparation of a pure electrolyte followed by electrowinning. The cemented copper is sent to smelting and refining. (These processes are discussed below.)

<u>Copper Sulfide Ores</u>. Copper sulfide concentrates are almost exclusively obtained by selective froth flotation by particles of copper minerals becoming attached to bubbles rising through a water-ore pulp to be collected as a concentrate froth. The non-copper minerals do not attach to the rising bubbles and they are discarded as tailings. The selectivity of the process is controlled by chemical reagents added to the pulp.

Beneficiation begins with crushing and wet-grinding of the ore (typically to 10-100 µm) such that the copper mineral grains are for the most part liberated from the gangue minerals. This is performed by a series of gyratory and cone crushers, and rod and ball mills.

Flotation is carried out immediately after grinding - in fact, some flotation reagents are added to grinding mills to ensure good mixing and a lengthy conditioning period. The flotation is carried out in cells whose principal functions are to provide: (1) clouds of air bubbles upon the surfaces of which the copper minerals attach and rise, (b) a mechanism for collecting the resulting concentrate froth from the cell, and (c) a means of underflowing the unfloated material into the next cell or to the waste tailings area.

Selective attachment of the copper minerals to the rising air bubbles is obtained by coating them with monolayers of collector molecules that have a sulfur atom at one end and a hydrocarbon "tail" at the other (e.g., sodium ethyl xanthate). Other important reagent are frothers (usually long-chain alcohols), which create a strong but temporary froth, and depressants (e.g., CaO, NaCN), which prevent the non-copper minerals from floating.

Flotation concentrates typically contain 20-30% copper. At this stage subsequent pyrometallurgical and electrolytic refining processes of copper concentrates are shared by secondary copper scrap depending upon concentrations and physical/chemical form of secondary material.

3.2 <u>Segregation of Copper Scrap</u>

Old copper scrap comes from a wide variety of sources, which include electric cables, piping, obsolete electronic/communications/process equipment, and various manufacturing operations.

Several routine techniques are traditionally use in the copper recycling industry to identify scrap for effective segregation. These include identification based on object recognition, color, apparent density, magnetic attraction, and chemical spot tests. These techniques, when properly applied by skilled sorters to mixtures of copper alloys, permit effective identification and segregation into specified categories (Riley 1983; Maynard 1972; Marr 1974).

Traditional sorting methods are often effective with regard to generic or descriptive specifications; however, there still can be many opportunities for introduction of impurities into alloys made from improperly segregated scrap. More sophisticated techniques have become commercially available and appear to offer improved product grade. These include fluorescent X-ray spectroscopy, portable optical emission devices, and thermoelectric sorters.

Copper scrap may be categorized into four main types on the basis of its copper content and the manner in which it is treated for copper recovery:

(1) Low-grade scrap of variable composition (10-95% Cu). This material is smelted in blast or hearth furnaces and then fire and electrolytically refined. It may also be treated in the Peirce-Smith converters of primary smelters.

(2) Alloy scrap, the largest component of the scrap-recovery system, consists mainly of brasses, bronzes, and cupronickels from new and old scrap. There is no advantage in re-refining these alloys to pure copper, and hence they are remelted in rotary, hearth, or induction furnace and recast as alloy stock. Some refining is done by air oxidation to remove aluminum, silicon, and iron as slag, but the amount of oxidation must be closely controlled because the desirable alloy constituents (Zn in brasses, Sn in bronzes) also tend to oxidize.

(3) Scrap, new or old, which is by and large pure copper but which is contaminated by other metals (e.g., metals used in plating, welding, or joining). This scrap is melted in the Peirce-Smith converters of primary smelters or the anode furnaces of primary or secondary refineries where large portions of the impurities (e.g., Al, Fe, Zn, Si, Sn) are removed by air oxidation. The metal is then cast into copper anodes and electrorefined as described below. It may also be sold as fire-refined copper for alloy making.

(4) Scrap which is of cathode quality and that requires only melting and casting. This scrap originates mainly as wastes from manufacturing (e.g., reject rod, bare wire, molds). It is melted and cast as ingot copper or alloyed and cast as brasses and bronzes.

3.3 Pyrometallurgical and Electrolytic Refining of Copper

Figure C-2 depicts principal processes employed in the recovery of copper from various grades of copper scrap. Secondary smelters use a three-stage smelting process that is equivalent to that employed in primary pyrometallurgical processing of mined copper ore: a first stage smelting process most commonly performed in a blast furnace, followed by a converter furnace, and an anode furnace. The final stage of copper refining involves an electrolytic process. Depending on grade, copper scrap may enter the flowstream at any of the three furnaces.

For select scrap, such as electric circuit board scrap, a hydrometallurgical process may be employed to produce chemical products such as copper sulfate and copper oxide. Each of the major elements of the recycling of copper scrap identified in Figure C-2 is described below.

3.3.1 Copper Smelting

Blast Furnace. The vertical shaft furnace, known as the blast-furnace and sometimes called the cupola, is the basic unit in a secondary copper smelter. Its ability to smelt copper-bearing material of an extremely diverse physical and chemical nature makes this furnace of prime importance in the smelting and refining process. It is the first unit that is employed in the pyrometallurgical treatment of low-grade secondary copper material and largely controls the metal losses in the system (Nelmes 1984).

The function of the blast-furnace is to economically smelt low-grade scrap, copper concentrates, and miscellaneous secondary materials. Whereas most scrap contains copper in elemental form, miscellaneous secondary material contains copper in the oxidic phase or in a powdery form that is accompanied by carbon-based material. When such secondary material is included in extractive metallurgy, blast furnace operation is considerably more complex. Their inclusion requires the melting process to be amended to reduce the oxidic portions of the furnace charge to an elemental form. Although secondary materials predominately consist of copper, other common constituents include iron, tin, lead, zinc, aluminum, nickel, arsenic, antimony, silica, chrome, magnesia, and various precious metals. Examples of secondary complex materials that may be processed in a blast furnace include:

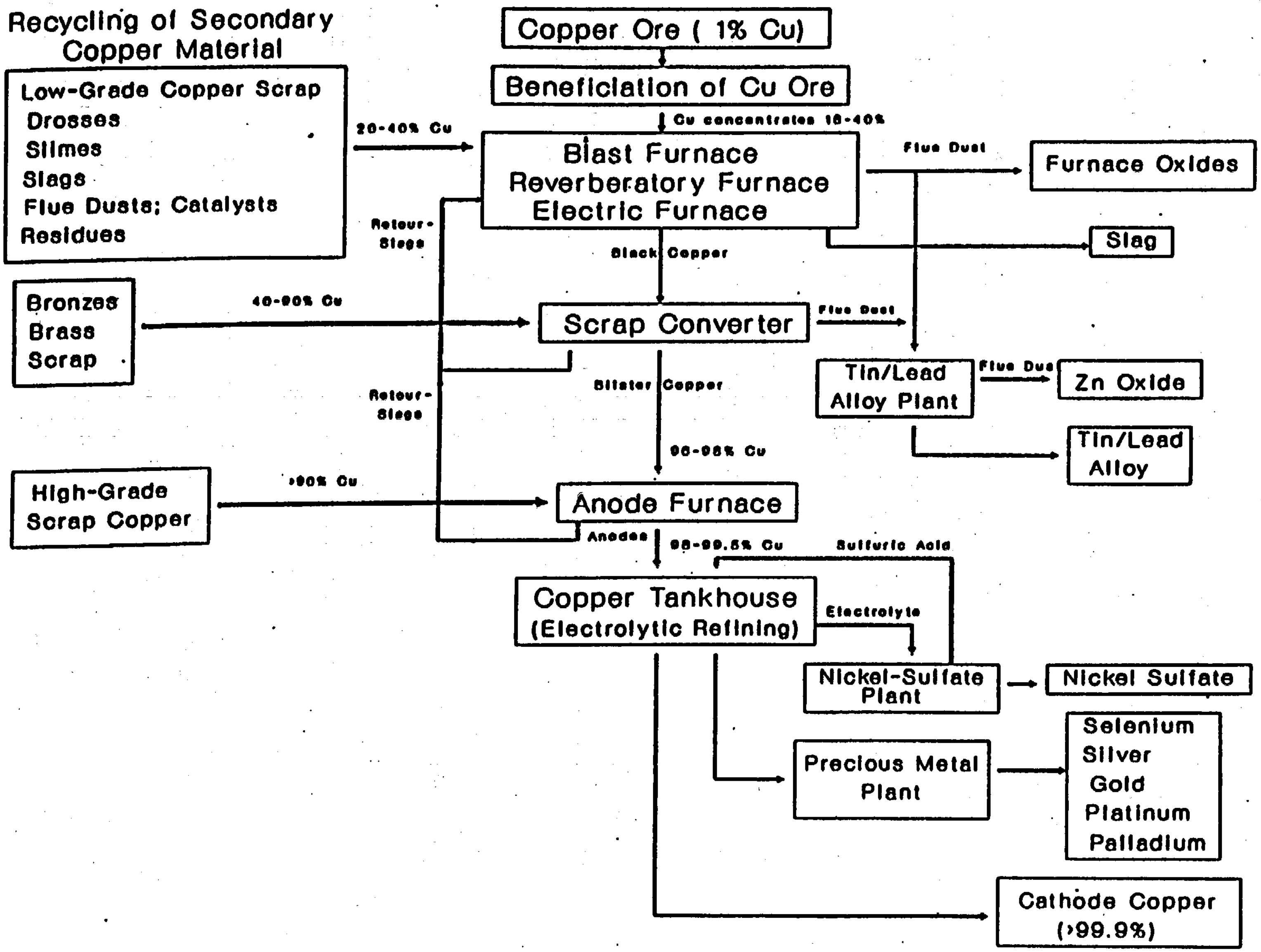

Figure C-2. Processing of Copper Containing Material

- Drosses - Drosses are materials that form on the surface of molten copper after contact with air, e.g. in a foundry. Drosses not only contain oxides but also drops of metallic particles, which are mechanically incorporated in the viscous oxide slag.

- Flue dust - During smelting and refining of primary and secondary copper, material is mechanically withdrawn from the gas phase and collected in the off-gas cleanup system, such as baghouses.

- Catalysts and collector dust - Some organic chemical reactions of industrial importance are catalyzed by copper compounds. After a certain operating time (months or years) the catalytical activity decreases and the copper catalysts have to be exchanged. Besides copper, the catalysts contain mainly carbon. Production waste from the manufacturing of collector dust also contains carbon and copper powder.

- Slimes from electroplating - Waste water from the electroplating process contains copper and other nonferrous metals. After precipitating with caustic soda or lime, the oxidic water containing slimes may be disposed or recycled.

- Metal rich slags - During the refining of copper alloy scrap by oxidation with air some copper is reported to the slag. Mainly the high copper content of converter slags justifies the installation of a reducing process stage in a secondary copper smelter.

- Copper cement - By cementation of copper solutions with iron scrap, copper precipitates as a mixture of an elemental and oxide powder.

Besides these materials there are many different residues, mainly with a low copper content. Depending on the copper price and their complexity, these materials can be recycled in an economic way or have to be passed on for waste disposal.

Furnace Design. Unlike iron blast-furnaces and iron melting cupolas, secondary copper blast-furnaces are rectangular in shape. This shape is necessary to allow penetration of the gas flow into the middle of the furnace. The width should not exceed 5 feet, thereby necessitating only a penetration of 2½ feet to the center. Increasing the size of the furnace necessitates an increase in the length. The size of the furnace is denoted by the cross sectional area at the tuyère level. Many furnaces are about 35 feet^2, although the largest is about 140 feet^2 and the smallest about 12 feet^2. Widths vary from 3.4 to 7 feet. It generally accepted that the height should be between 12 and 15 feet (Nelmes 1984).

The vertical shaft furnace contains water jackets which extend from the hearth section to the charge door level (Figure C-3). The water jackets are usually set in tow tiers and are strapped together to reduce warping. Approximately 100 gallons per square foot per hour water flow is adequate to keep the jacket cool and to form slag lining on the inner face of the jacket. This lining protects the jacket from the high-temperature oxidizing gases, metal, and slag.

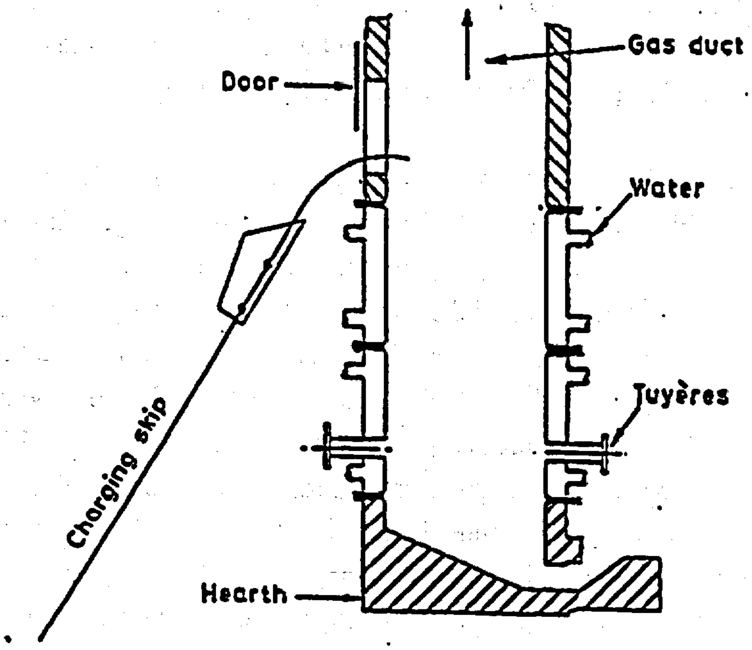

Figure C-3. Secondary Copper Blast-Furnace

A 30-foot header tank is used to store sufficient water to safeguard the jackets in the event of power failure to the pumps. It also provides a constant head of water so that pressure variations do not occur.

The hearth of the furnace is a rectangular box or crucible attached to the lower ends of the jackets and lined with chrome-magnesite refractory. It is constructed as a slope that extends from the sides of the crucible towards the tap hole. The accumulation of liquid metal and slag on the hearth is very small, being only sufficient to cover the gas-trap type of taphole.

The size of the furnace is obviously governed by the input capacity required and the rate of operation. The rate is quantified by the amount of copper-bearing material smelted per square foot of cross-sectional area at tuyère level per day. The rates may vary between 4 and 8 t/ft^2 per day. If it is desired to smelt 240 t/day at a rate of 6 t/ft^2 per day, obviously, the furnace will be 40 ft^2. As the width should not exceed 5 feet and, in this case, may be 4 feet, the furnace would be 10 feet long by 4 feet wide and about 14 feet high.

Furnace Operation. Blast furnace feed is generally prepared by pelletizing or briquetting finely divided material (Browne 1990). The scrap charge is fed onto a belt conveyors, which in turn discharges into one of two skip hoist buckets. These in turn are hoisted and alternately dumped into opposite sides of the furnace. As reducing agent, coke is used and as a flux, silica, lime, or iron oxide is generally added. Air is injected into the furnace by means of tuyères situated uniformly around the shaft. The combustion of coke in vertical shaft furnaces is complex and has been thoroughly described (Evans 1964; Breen 1963). The amount of coke required per ton of charge is related to both the heat required for melting and the amount of oxide in the charge that must be reduced to elemental metal and sufficient flux must be added to provide a "fluid" slag. The blowing rate and the oxygen enrichment level must be matched to the furnace charge rate. The copper-bearing material initially enters at the top of the furnace into a zone at 400-600°C. It subsequently descends into the tuyère zone and increases in temperature to about 1,400°C (Schwab 1990).

The blast furnace crucible is continuously tapped through a liquid seal to prevent the blast air from blowing out the bottom. A mixture of molten copper and slag flows down a launder into an oil-fired rocking furnace that can rotate through an approximately 120 degree arc. This furnace is large enough to give the slag sufficient time to separate from the copper. Rotating the furnace in one direction allows the liquid copper to fill a preheated ladle on a rail car below the rocking furnace. Rotation in the opposite direction allow the slag to pour into a granulating trough. Granulation is accomplished by hitting the liquid slag stream with a high pressure jet of water. The slag and water are collected in a pit that is large enough to remove the slag with a clambell bucket on a crane. The liquid metal in the ladle, known as black copper, is approximately 80% copper.

Blast Furnace Yields. Due to the variability of feed materials and operating parameters of a blast furnace, metal recoveries and yields vary. Typical analyses of outputs from blast furnaces in which the feed is secondary copper scrap are shown in Table C-7.

Table C-7. Typical Composition of Principal Outputs from Secondary Blast-Furnace Operations

Percent

	Cu	Sn	Fe	Zn	Pb	Ni	Al_2O_3	CaO	SiO_2	Other
Black Copper	80	4	5	3	4	4	—	—	—	<1
Furnace Oxides	1.5	1	—	50	15	—	—	—	—	~32.5
Final Slag	0.9	0.3	30	3	0.6	0.15	9	14	27	~15

Source: Nelmes 1984.

In turn, the total metal content of black copper typically represents about 40% of the charge weight with slag and furnace oxide representing 40% and 5%, respectively.

Table C-8 shows the corresponding quantities of major constituents in the blast furnace output that corresponds to a furnace charge of 100 tons, and Table C-9 provides corresponding values for metal recovery.

Table C-8. Typical Constituent Quantities of Black Copper Furnace Oxides and Slag Corresponding to a 100 Ton Blast-Furnace Charge

Quantities (tons)

Furnace Output	Cu	Sn	Fe	Zn	Pb	Ni	Al_2O_3	CaO	SiO_2	Other
40 t Black Copper	32.0	1.6	2.0	1.2	1.6	1.6	—	—	—	—
5 t Oxides	0.08	0.05	—	2.5	0.75	—	—	—	—	1.6
40 t Slag	0.36	0.12	12.0	1.2	0.24	0.06	3.6	5.6	10.8	6.0
Total	32.44	1.77	14.0	4.9	2.59	1.66	3.6	5.6	10.8	7.6

Source: Nelmes 1984

Table C-9. Percent Recovery Under Typical Operating Conditions

Output	Cu	Sn	Fe	Zn	Pb	Ni	Al_2O_3	CaO	SiO_2
in Metal	98.64	90.4	14.29	24.49	61.78	6.39	—	—	—
in Oxide	0.25	2.82	—	51.02	28.96	—	—	—	—
in Slag	1.11	6.78	85.71	24.49	9.26	3.61	100	100	100
Total	100	100	100	100	100	100	100	100	100

Source: Nelmes 1984

Blast Furnace Slags. The function of the slag is to carry away otherwise infusible material and is composed mainly of calcium, iron, and silicon. When granulated blast furnace slag is dried, crushed, and screened, it is used to produce a variety of commercial products. Slag granules are hard, dense, and inert; and based on particle sizing, they are ideal for making a variety of abrasives, a filler of asphalt shingles, roofing sealers, road surface bedding, grit for sand-blasting, and the manufacturing of mineral wool and light-weight cement/concrete (Nelmes 1984; Schwab 1990; Mackey 1993).

Reverberatory Furnaces. Reverberatory furnace smelting began in the nineteenth century and still accounts for a significant fraction of both primary copper production and recycling of secondary scrap material. The disadvantages of these furnaces, however, are the long melting cycle times and their low fuel efficiencies (Davenport 1986).

In a reverberatory smelter furnace the copper scrap is charged into one or more scrap pile located behind each other in front of several high capacity end-wall fired burners. These high capacity conventional burners typically are fired above the copper scrap and use the reverberatory effect for heat transfer, i.e., re-radiation from the refractory roof and walls to the scrap (Thomas 1991). During the melting cycle, when the process requirement for energy are high, the surface area of scrap exposed to the flame radiation and to radiative heat transfer from the furnace refractory surfaces is low relative to the total surface area of the scrap. This is due to the top layers of scrap shading the interior scrap surfaces from direct radiation resulting in low rates of radiative heat transfer to the entire scrap. In addition, convective heat transfer to the interior of the scrap charge is low due to the low velocity burner designs resulting in limited flame momentum to penetrate the scrap pile, therefore, resulting in limited circulation of gases within the scrap.

During the scrap melting process, the temperature is continuously increasing while the surface area is reducing via partial melting down. This dynamic change in heat sink characteristics makes it necessary to change flame characteristics throughout the entire melt down cycle to maximize heating efficiency and productivity.

A typical reverberatory furnace is charged with approximately 250 tons of scrap and about 100 tons of liquid metal in order to maintain a 24-hour operating cycle, with the melting portion of the cycle of 8 hours (Table C-10). This represents an average "melt-in" rate of cold scrap of about 31 tons per hour.

Table C-10. Operating Parameters Typical of a Large Reverberatory Smelting Furnace

Metal Charged, (short tons)	
Cold Scrap	250
Molten Metal	100
Melt-In Cycle (hours)	8
Melt Rate (short tons/hr)	31
Total Cycle (hours)	24
Daily Production (tons/day)	350

Source: Wechsler 1991

Electric Furnaces. Some copper smelting is carried out in electric furnaces. These are similar to reverberatory furnaces, but are powered by six self-baking carbon electrodes passing electrical current through the slag and matte layers. The advantages of electric furnace smelting are a small off-gas volume, a discardable low-copper slag, and a clean operation. However, electric furnaces make only limited use of the energy from sulfur and iron oxidation and they are expensive to build and operate. They have found only limited use for secondary copper smelting (Davenport 1986).

3.3.2 Converting Copper

The black copper liquid matte produce by the smelting furnace contain significant amounts of iron, tin, lead, zinc, and nickel metals, a well as sulfur, which must be removed by a converter furnace. Copper alloy scrap, such as brass, bronze, and German silver, may also be processed in converter furnace since these alloys also contain amounts of zinc, tin, and nickel. In a converter furnace, these elements are removed either by reduction and evaporation or by oxidation. Tin is recovered from baghouse dust as tin/lead alloy used for soldering, and zinc is converted to zinc oxide for the pigment industry (Glockman 1992).

For the converting process of black copper or copper alloy scrap either a Peirce-Smith converter or a Top Blown Rotary Converter (TBRC) are used. In converter furnaces, oxygen-enriched air or pure oxygen is used for the successive removal of secondary metals (Davenport 1986; Roscrow 1983)

A schematic of the TBRC furnace is shown in Figure C-4. A process cycle begins by tilting the furnace to the vertical position for charging. Furnace feed, which consists of copper alloys (brass or bronze) or black copper, is hoisted in stages until the entire charge is added. The charge is then melted with the burner operating under reducing conditions to prevent premature oxidation of metallic copper. During this period the furnace is only rotated intermittently.

When the whole charge is molten, the cycle commences by increasing the furnace rotation capacity. During the smelting cycle, the iron reductant reduces copper oxides to metallic copper and at the same time some of the lead, tin, and zinc oxide are also reduced to metals. However, the bulk of the zinc escapes as fumes and is collected as a zinc-rich dust in a baghouse.

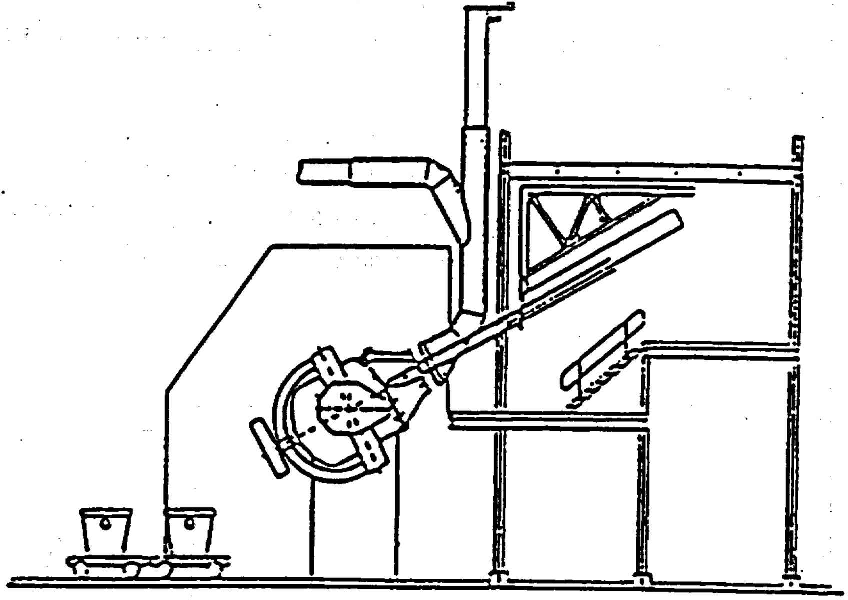

Figure C-4. Schematic of a TBRC Plant (from O'Brien 1992)

The bulk of the iron formed reacts with silica flux to form a slag that typically contains 0.5-1.0% copper. At the end of the smelting cycle, the furnace is titled down to pour the slag off.

The slag can either be granulated and sold as shot blasting grit or used for fill. The black copper remaining in the furnace is saturated with iron (up to 7%) and typically contains up to 10% of lead, zinc, and tin.

The furnace is then returned to the operating position and the speed increased to 20 rpm to start the refining cycle by blowing air or oxygen enriched air onto the surface of the agitated melt. The remaining impurities are removed in the order of iron, zinc, tin, and lead and a lead/tin-rich fume is collected in a secondary baghouse. Extra silica is added to dissolve any iron formed and a secondary copper-rich slag is also produced. This slag can be left in the furnace to form part of the next charge; or if the impurity levels, such as nickel, are too high, it is removed from the converter for separate treatment.

Operating parameters and total cycle time is dependent upon the composition of furnace feed and the desired output product. The refining cycle can be controlled to produce various copper alloys with significant residual quantities of secondary metals or continued until successive oxidation and removal of metals results in <u>blister copper</u> (96-98% Cu), which is cast into molds and subject to further refining.

Annual throughput of a TBRC will vary according to the average copper content in the furnace feed and the desired final product. Figure C-5 identifies representative throughputs of recycled copper alloys or blister copper as a function of furnace size (O'Brien 1992).

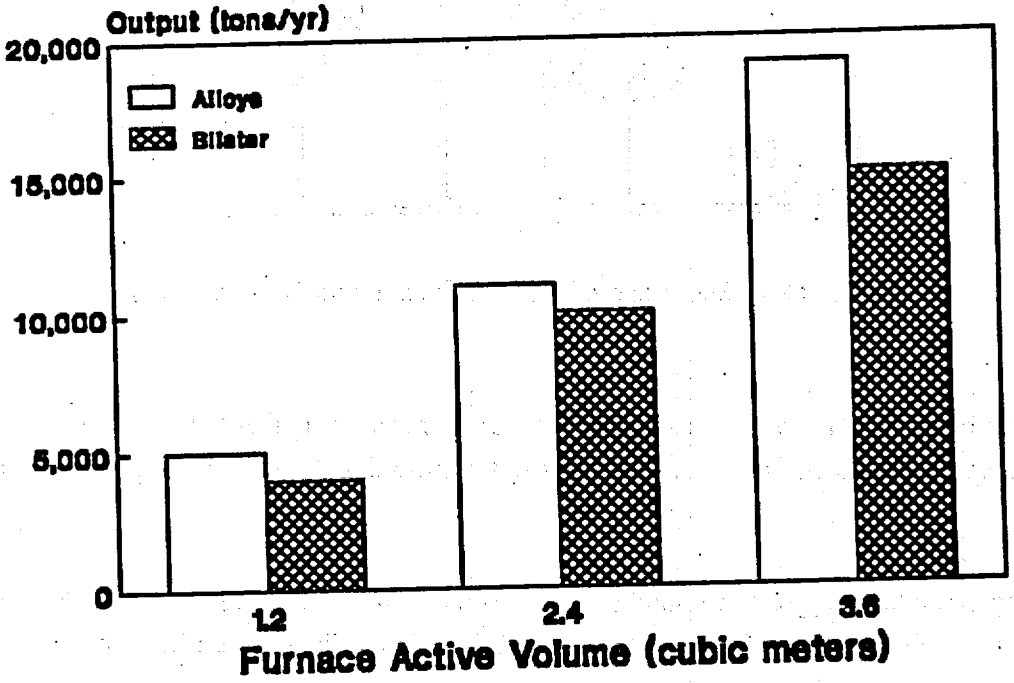

Figure C-5. Annual Output of Product

<u>Converter Effluents</u>. The volume of gases leaving rotary converter furnaces is relatively low since oxygen-enriched to nearly pure oxygen is used for combustion and oxidizing metallic impurities contained in the black copper feed material. Hoods are used to collect fugitive emissions at the charging and tapping locations to control ambient plant environments.

Converter furnace effluent gases are cooled by dilution with air, which is drawn from around the furnace and air from charging and tapping hoods before subjected to a baghouse filtration system. Based on metal content, the collected filtrate is shipped to zinc smelters or tin and lead refiners for metal recovery (George 1993).

3.3.3 Copper Anode Production for Electrorefining

The blister copper produced by smelting and converting contains about 99.5% Cu, 0.5% O, and 0.05% S plus small amounts of impurities such as As, Bi, Fe, Ni, Pb, Sb, Se, Te, and precious metals (Davenport 1986). The production of copper anodes is the last step in the pyrometallurgical process and is commonly referred to as "fire refining." Almost all copper is destined for electrolytic refining; hence, it must be cast into strong flat anodes. Sulfur and oxygen are removed prior to the casting to avoid formation of SO_2 blisters. The sulfur is removed by air oxidation and the oxygen by reduction with hydrocarbon gases, usually in a cylindrical anode furnace.

The anodes for electrorefining are most often cast in copper molds on rotating horizontal wheel. The mold shape includes lugs by which the anodes are supported in the electrolytic refining cells. In modern plants, the thickness of the anodes is controlled precisely by continuously weighing the quantity of copper being poured into the molds. This ensures that each anode has the same lifetime in the electrorefining cell.

3.3.4 Electrolytic Refining

The final stage in copper purification that yields copper with less than 40 ppm impurities (i.e., less than 0.04 g per kilogram) and with a controlled oxygen content (0-0.03%) involves an electrolytic process conducted in a copper tankhouse.

Electrorefining consists of placing copper anodes and pure-copper cathode starting sheets in a $CuSO_4$-H_2SO_4-H_2O electrolyte and passing a direct electrical current between them. The electrical current causes copper to dissolve from the anode and to be plated in pure form on the cathode. About 0.25 V is required. One or more organic "smoothing agents" are added to the electrolyte to enure that the newly plated copper is dense and smooth.

More modern copper electrorefining facilities today have replaced the traditional copper cathode starter sheets with reusable stainless-steel cathode plates. Stainless-steel starter cathodes are straighter and can, therefore, be hung closer together. This permits use of high current densities, which in turn yields a higher purity of copper cathode (Suttkill 1992).

A large modern copper tankhouse facility may consist of 50 cells each containing 56 cathodes and 57 anodes. The cells are constructed of reinforced concrete with wooden bottoms and lead linings. Cell dimensions are about 5.6 meters in length, 1.2 meters in width, and 1.4 meters in depth to provide a submerged cathode area of 1.2 square meters.

In industrial practice, the cathodes are grown for 10-14 days, after which time they weigh about 150 kg. They are then removed from the cells and replaced by new starting sheets. Anodes remain in the electrolytic cells until they are almost completely dissolved, usually for two batches of cathodes. They, too, are then removed and replaced by a new set of anodes. The undissolved anode scrap is washed, melted, and recast as fresh anodes for further refining.

The prescribed high purity of the cathode copper is obtained by controlling the purity of the electrolyte and by making sure that the whole plant is operated under carefully controlled conditions. The impurities coming into the refinery in the anodes fall into two categories (Davenport 1986):

(1) those which do not dissolve and which consequently can be removed from the electrolytic cell in solid form after a refining cycle, for example, Au, Pt metals, Ag (precipitated by the small additions of Cl^- to the electrolyte), Pb, S, Se, and Te; and

(2) those which dissolve to an appreciable extent from the anode and which must be removed from the electrolyte to avoid build-up and eventual contamination of the cathodes, for example, As, Bi, Co, Fe, Ni, and Sb.

The soluble impurities are removed by continuously bleeding a portion of the electrolyte through a purification system in which they are precipitated electrolytically or by evaporation. Many of the impurities are sold as byproducts either before or after further treatment. Particularly important among these are gold and silver, though the Pt-group metals, As, Bi, Co, Ni, Sb, Se, and Te, may all be sold in one form or another.

3.3.5 Melting, Casting, and Use of Cathode Refined Copper

The cathodes from the electrolytic cells are washed then melted and cast into shapes for fabrication and use. The melting is usually done in gas-fired ASARCO vertical shaft furnace (Davenport 1986). Stacks of cathodes are charged near the top of the furnace to be melted as they descend, heated by rising combustion gas. The melted copper flows continuously from the furnace to be held in a gas-fired or induction-heated rotary holding furnace before casting. It is most often cast and rolled continuously to form copper rod for wiremaking. Alternatively, it may be cast as slabs or billets for mechanical use or as individual wirebars for rolling and drawing into wire. Other important grades of copper are oxygen-free copper (<0.001% O) for electronic use, and phosphorus-deoxidized copper for applications involving welding.

The most important properties of the final copper product are its electrical conductivity and its mechanical behavior during fabrication and use. These are both adversely affected by impurities in the copper, particularly Bi, Se, Te, As, Sb, and S. The product copper is analyzed routinely for these elements to ensure that proper refining, melting, and casting procedures are being carried out. In addition, conductivity, hardness, tensile strength, and springback tests are performed to ensure that the copper is of good physical quality.

3.4 Recycling of High-Grade Copper by Non-Pyrometallurgical Methods

The wire and cable sectors of various industries generate considerable amounts of electrical scrap. The copper in cable scrap is generally of high grade and free of significant amounts of metal and non-metal contaminants.

Conventional recycling of high-grade scrap requires at least one melting operation in a furnace. The Conform extrusion method is a process that directly recycles granulated copper feedstocks into a variety of finished products (Hordyk 1994).

Feedstock preparation includes granulation of wire scrap by a rotating blade or hammermill to produce uniform copper granules. Separation of non-metallic constituents, such as polyethylene or polyvinyl cellulose insulation, is accomplished by means of a fluidized air bed separator and ferrous particulates are removed magnetically. Any residual volatile contaminant, such as grease or oils, are removed by heating in a furnace charged with a reducing gas to ensure that the granules are not oxidized.

With proper preparation of feed material, the physical process of converting scrap by conform extrusion to a finished product is a straight forward process. The extrusion process consists of a rotating wheel that has a circumferential groove - this acts as a chamber/metal reservoir which is necessary to make the process continuous. A quandrant of the circumference of the wheel is kept in close contact with a stationary steel shoe which incorporates the tooling for extrusion (Figure C-6).

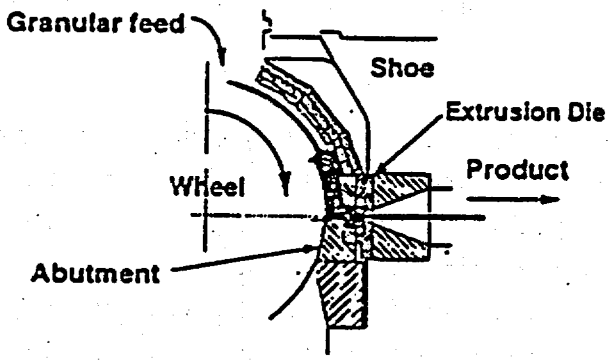

Figure C-6. Conform Extrusion Process

Feedstock is introduced into the groove and is drawn into the extrusion chamber by the frictional grip. The forward motion of the feed is stopped by a solid metal block, known as the abutment, which is inserted into the groove. At this point, the feedstock yields axially and the consequent frictional forces, due to continued wheel rotation, generates the temperature and pressure required for extrusion to occur.

It is the high temperature and pressure that enables the discreet metal granules to weld together and thus continuously extrude the solid product. From this simple principle, high quality solid and hollow section products have been manufactured to international standards from range of nonferrous materials.

Product output using a standard Holton C300H machine with a 13-mm grove width are in excess of 500 kg/hr. Development is underway to extend this to 700 kg/hr. Some of the products that can be Conform extruded are:

- Profiles - The Conform is especially suited for the manufacture of annealed strip and shaped magnet wire profiles, which are difficult to produce by conventional methods of production. Different shapes are made to high standards of finish and tolerance by simply changing the extrusion die.

- Re-draw Rod - Granule can be processed into annealed re-draw rod of any conventional size between 3.5-mm and 8-mm in diameter meeting the relevant international product standards. Provided that the granules have been adequately cleaned, the rod can be redrawn on conventional high speed equipment. However, re-draw to fine wire is not recommended.

- Soudronic Welding Wire - Soudronic AG specify particular properties for the welding wire used in their machines. There are numerous types of Soudronic welding machines which require different wire mechanical properties depending on the age of the machine and the user preference. In most cases, wire extruded directly via Conform will not achieve these properties, thus further drawing is necessary to meet the specification. However, producing re-draw rod of 8-mm diameter with subsequent high speed drawing to the correct Soudronic wire diameter is the most economical production route.

- Annealed Wire - Annealed wire can be directly produced from copper granules by Conform. A minimum size of 2-mm diameter is recommended for economical reasons. However, extremely high pressure, high flash rates, and lower output rate are associated with directly extruding wire. Thus although directly extruding wire is possible, it is not generally recommended on a cost effective production route.

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APPENDIX D

SELECTION OF RADIONUCLIDES FOR RADIOLOGICAL IMPACTS ASSESSMENT

Table of Contents

List of Tables

SOURCES USED TO MAKE RECOMMENDATIONS

The following sources were reviewed and used to arrive at the recommendations as to which long-lived (*i.e.*, half-lives greater than six months) radionuclides should be included in the analysis of the potential for recycling scrap metal from nuclear facilities. The nuclides selected from each source and considered as candidates for the analysis are listed in Table D-6. Each source is referred to by an abbreviation, which in most cases is the document number.

IAEA 95. Table I of IAEA 95 presents clearance levels—expressed in units of Bq/g—for the unconditional release of material contaminated with radioactivity. To determine these levels, the IAEA reviewed a large number of documents. Specific to the recycle of metals (including steel, aluminum and copper), the IAEA reviewed the following four documents: IAEA 92, CEC 88, Elert 92 and Garbay 91. The radionuclides that each of these four documents included in their analyses of recycling impacts (along with release limits) are tabulated on Table I.3 of IAEA 95. Only those radionuclides that are associated with metal recycle are considered as candidates for the scrap recycle analysis.

NUREG/CR-0134. In *Potential Radiation Dose to Man from Recycle of Metals Reclaimed from a Decommissioned Nuclear Power Plant*, NUREG/CR-0134, O'Donnell *et al.* present individual and population dose factors resulting from scrap recycle for 27 radionuclides. These radionuclides "... include fission and activation products (except gaseous species) that may be encountered during decommissioning, and that have radioactive half-lives longer than about 40 days, ^{239}Pu and ^{241}Am (to characterize transuranic contaminants), and ^{234}U, ^{235}U, and ^{238}U."

WINCO-1191. The radionuclides reported in *Radionuclides in the United States Commercial Nuclear Power Reactors*, WINCO-1191 (Dyer 94) were taken from a study of pipe samples and pipe surface contamination from pressurized and boiling water reactors, and are shown on Table D-1. The samples were from 11 pressurized water reactors and "over" eight boiling water reactors. The data were based on surface samples taken from the inside of stainless steel piping, a main coolant system check valve and from fuel element hardware. The study also includes an analysis of Shippingport reactor material samples. Radionuclides that are found exclusively in the coolant or within the fuel cladding are excluded from the scrap recycle analysis.

Table D-1: Nuclides from WINCO-1191

Nuclide	Half-Life (yr)	Specific Activity at Shutdown (μCi/cm^2)
C-14[a]	5.73E+03	< 5.9E-8
Mn-54[a]	8.55E-01	6.9E-3
Fe-55[a]	2.73E+00	2.7
Co-57[b]	7.44E-01	1.78E-5
Ni-59[a]	7.50E+04	6.8E-3
Co-60[a]	5.23E+00	2.0
Ni-63[a]	1.00E+02	1.55
Zn-65[b]	6.68E-01	1.68E-6
Nb-93m[a]	1.46E+01	1.2E-2
Nb-94[a]	2.00E+04	8.4E-5
Ag-110m[b]	6.84E-01	1.3E-4
Mo-93[c]	3.50E+03	1.8E-8 μCi/g
Sb-125[c]	2.73E+00	1.0E-5 μCi/g
I-129[a]	1.57E+07	<1.6E-8
Ce-144+D[b]	7.80E-01	2.49E-6
Pu-238[a]	8.77E+01	1.2E-7
Pu-239/ 240[a]	6.56E+03 (Pu-240)	4.7E-8
Cm-244[a]	1.81E+01	2.6E-8

[a] Sample taken from Shippingport B-loop Primary Coolant Check Valve. Total activity of sample 6.27 μCi/cm^2.

[b] Sample taken from Ranch Seco Nuclear Power Plant. Total activity is 0.252 μCi/cm^2.

[c] Sample taken from Shippingport reactor internals. Activity is in μCi/g. Total activity of sample was 3.85E-3 μCi/g.

The study notes that between 86% and 99% of the radioactivities from the pipe walls and pipe surfaces are the activation products Fe-55, Co-60 and Ni-63. Dyer *et al.* go on to note that the distribution of radionuclides for reactor component appears to be the same whether the activities are on surfaces or are within the part materials.

NUREG/CR-0130. Appendix J of *Technology, Safety and Costs of Decommissioning a Reference Pressurized Water Reactor Power Station,* NUREG/CR-0130 (Smith 78) presents five sets of "reference radionuclide inventories" that were used to characterize a PWR at the time of its decommissioning. Four of the reference radionuclide inventories are associated with contaminated metal components, and are given on Table D-2, while the fifth set is for contaminated concrete, and is not related to this study.

The metals being removed during PWR decommissioning which are contaminated with either activated corrosion products or surface contamination would definitely be candidates for recycling. Smith 78 includes the "stainless and carbon steel activation products" classes of radionuclides, which are the contaminants on the reactor vessel and its internals. In a PWR at the time of its decommissioning, this metal would be too highly activated to be a candidate for recycling. However, stainless and carbon steel can become activated by other means, or a reactor may have operated for only a short time (*e.g.*, Shoreham), therefore, the radionuclides in these two sets should be included in the scrap recycle analysis.

Konzek *et al.* (Konzek 93) revised the PWR decommissioning analysis originally presented in Smith 78 to reflect current regulations, practices and costs. The authors did not re-analyze the radiological source terms presented in Appendix C of Smith 78, although they did use "as built" drawings, rather than design drawings, for estimating the volume of waste material and equipment.[1] This could change the radionuclide inventories but would not result in any major changes to the expected radionuclide distributions in PWR components at the time of decommissioning.

NUREG/CR-3585. In *De Minimis Impacts Analysis Methodology*, NUREG/CR-3585, (Oztunali 84), the authors present an analysis of the impacts of metal recycling. Any metal which met the *de minimis* radionuclide level would have been considered to be a candidate for recycling, since it would no longer have been under regulatory control.

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[1] M. Bierschbach (Pacific Northwest Laboratory), Private communication 7/17/96.

Table D-2: Nuclides Included in NUREG/CR-0130

Nuclide	Stainless Steel AP	Carbon Steel AP	Activated Corrosion Products	Surface Contamination
Mn-54	✓[a]	✓	✓	✓
Fe-55	✓	✓	–	✓
Co-60	✓	✓	✓	✓
Ni-59	✓	✓	–	–
Ni-63	✓	✓	–	–
Zn-65	✓	–	–	–
Sr-90	–	–	–	✓
Mo-93	✓	✓	–	–
Nb-94	✓	–	–	–
Ru-106	–	–	✓	–
Cs-134	–	–	–	✓
Cs-137	–	–	✓	✓

[a] A check mark (✓) indicates that the radionuclide is included in the NUREG/CR-0130 reference inventory.

NUREG/CR-4370. *Update of Part 61 Impacts Analysis Methodology*, NUREG/CR-4370, (Oztunali 86) was reviewed as a source of information concerning the radiological profile of scrap which would be disposed of as low-level waste—recycled scrap would have a similar profile. The report analyzed 53 radionuclides, up from the 23 analyzed in the original Part 61 analysis methodology. Table D-3 presents the 53 radionuclide that were analyzed in NUREG/ CR-4370.

Oztunali 86 identifies 148 waste streams, for which it develops radionuclide characterizations. Only a few of the 148 waste streams are directly applicable to the recycling of scrap. The waste streams which are applicable to scrap recycle are those associated with:

Table D-3: Nuclides Analyzed by NUREG/CR-4370

Nuclide	Notes	Nuclide	Notes	Nuclide	Notes
H-3	a, b, c	Cs-135	a, b, c	U-236	c
C-14	a, b, c	Cs-137	a, b, c	U-238	a, c
Na-22	NI	Eu-152	b	Np-237	a, b, c
Cl-36	-	Eu-154	b	Pu-236	c
Fe-55	a, c	Pb-210	NI	Pu-238	a, b, c
Co-60	a, c	Ac-227	HLW	Pu-239	a, b, c
Ni-59	a, c	Th-228	-	Pu-240	a, c
Ni-63	a, b, c	Th-229	NI	Pu-241	a, b, c
Sr-90	a, b, c	Rn-222	NI	Pu-242	a, b, c
Nb-94	a, c	Ra-226	-	Pu-244	NI
Tc-99	a, b, c	Ra-228	NI	Am-241	a, b, c
Ru-106	b	Th-230	HLW	Am-243	a, b, c
Ag-108m	NI	Th-232	NI	Cm-242	b, c
Cd-109	NI	Pa-231	HLW	Cm-243	a, b, c
Sn-126	b	U-232	HLW	Cm-244	a, b, c
Sb-125	b	U-233	-	Cm-248	HLW
I-129	a, b, c	U-234	c	Cf-252	HLW
Cs-134	b	U-235	a, c		

a. Associated with the nuclear power plant decommissioning contaminated metals waste streams.
b. Associated with the West Valley Demonstration Project equipment and hardware waste streams.
c. Associated with non-compressible trash waste streams.
NI Nuclide was not included in the characterization of any of the waste streams in NUREG/CR-4370. May be included as a decay product of another nuclide which is included in the waste stream characterization.
HLW Nuclide was only included in the spent fuel reprocessing high-level liquid waste waste stream.

1. The nuclear power plant decommissioning contaminated metals waste streams

2. The West Valley Demonstration Project equipment and hardware waste streams

3. Non-compressible trash waste streams

SAND92-0700. In volume 3 of the *Preliminary Performance Assessment for the Waste Isolation Pilot Plant, December 1992,* SAND92-0700/3 (SNL 92), A. Peterson estimates the radionuclide inventories in DOE-generated transuranic (TRU) waste that would be disposed of at the Waste Isolation Pilot Project (WIPP). Because the radionuclides present in transuranic waste are a likely source for the contamination of metals present at DOE facilities, Peterson's memo is included in the present review. The memo classified TRU waste as to whether it can be contact handled, or whether remote handling is required. Both types of TRU waste are considered for the scrap recycle analysis, and Table D-4 indicates the type of TRU waste that the radionuclide may be expected to be found.

ORIGEN. The Oak Ridge Isotope GENeration and depletion code (ORIGEN, Croff 80) has approximately 1700 nuclides in its library, collected into three groups: activation products, transuranics and fission products. Of these, there are 1040 unique, non-stable nuclides, but only 127 of these have half-lives greater than six months. (Note, a given nuclide can appear in more than one of ORIGEN's three nuclide groupings.)

To determine which of these 127 radionuclides should be included in the scrap recycle analysis, an ORIGEN analysis was performed to calculate the activity in spent fuel at the time of its discharge from the reactor. An initial enrichment of 3.04% was assumed, with a burnup of 44,340 megawatt-days per metric ton of initial heavy metal (MWD/MTIHM) and the characteristics of PWR fuel with impurities. The activities were combined with the dose factors from Federal Guidance Reports (FGR) No. 11 (Eckerman 88) and 12 (Eckerman 93) in the following manner:

$$R_i = \frac{A_i \cdot DCF_{j_i}}{A_{max} \cdot DCF_{j_{max}}}$$

where:

R_i = ratio for radionuclide i

A_i = spent fuel activity for radionuclide i

DCF_{j_i} = dose conversion factor for pathway *j* of radionuclide *i*

A_{max} = spent fuel activity for the radionuclide with the maximum product of activity and dose conversion factor for pathway *j*

$DCF_{j_{max}}$ = dose conversion factor for the radionuclide with the maximum product of activity and dose conversion factor for pathway *j*

Three ratios were calculated for each of the 127 radionuclides, corresponding to the three dose pathways: inhalation, ingestion and external exposure. The maximum dose conversion factors for inhalation and ingestion of each nuclide were taken from FGR 11, while the dose coefficients for external exposure to soil contaminated to an infinite depth from FGR 12 were used to characterize external exposure. It was found that Cs-134 gave the maximum product of activity and external exposure dose coefficients (*i.e.*, $A_{max} \times DCF_{j_{max}}$), while Cm-244 gave the maximum product of activity and dose conversion factor for both inhalation and ingestion.

The results of this effort are shown on Table D-5. Any radionuclide with any of its three ratios greater than 10^{-4} is considered to be a candidate for the scrap recycle analysis, and is indicated by a check mark (✓) in the "ORIGEN" column of Table D-6.

SAND91-2795. The *Yucca Mountain Site Characterization Project, TSPA 1991: An Initial Total-System Performance Assessment for Yucca Mountain*, SAND91-2795 (Barnard 92) presents an analysis of the impacts from the disposal of spent fuel. Because the radionuclides present in spent fuel are a likely source for the contamination of metals present in nuclear power plants and other tail-end fuel cycle facilities, this report was included in the present review.

Table D-4: Nuclides Analyzed by SAND92-0700 for WIPP

Nuclide	Half-Life (yr)	Remote Handled	Contact Handled	Nuclide	Half-Life (yr)	Remote Handled	Contact Handled
Mn-54	8.56e-01	✓	–	Th-232	1.41e+10	✓	✓
Co-60	5.27e+00	✓	–	U-233	1.59e+05	✓	✓
Ni-63	1.00e+02	✓	–	U-235	7.05e+08	✓	✓
Sr-90	2.91e+01	✓	✓	U-236	2.34e+07	✓	–
Tc-99	2.13e+05	✓	–	U-238	4.47e+09	✓	✓
Ru-106	1.01e+00	✓	✓	Np-237	2.14e+07	✓	✓
Sb-125	2.77e+00	✓	–	Pu-238	8.77e+01	✓	✓
Cs-134	2.06e+00	✓	–	Pu-239	2.41e+04	✓	✓
Cs-137	3.00e+01	✓	✓	Pu-240	6.56e+03	✓	✓
Ce-144	7.78e-01	✓	✓	Pu-241	1.44e+01	✓	✓
Pm-147	2.62e+00	✓	✓	Pu-242	3.75e+05	✓	✓
Eu-152	1.33e+01	✓	–	Am-241	4.33e+02	✓	✓
Eu-154	8.80e+00	✓	–	Cm-244	1.81e+01	✓	✓
Eu-155	4.96e+00	✓	–	Cf-252	2.64e+00	✓	✓

Table D-5: Nuclides from ORIGEN with Their Normalized Activity Weighted Dose Factors

Nuclide	Ground	Inhalation	Ingestion	Nuclide	Ground	Inhalation	Ingestion
H-3	0.00e+00	3.04e-08	2.31e-06	Rh-102	1.16e-05	1.27e-07	8.42e-07
Be-10	2.96e-15	1.16e-12	1.15e-12	Pd-107	0.00e+00	1.09e-09	9.72e-10
C-14	3.95e-12	7.27e-10	5.51e-08	Ag-108m	6.40e-08	2.23e-09	4.54e-09
Na-22	0.00e+00	0.00e+00	0.00e+00	Ag-110m	6.04e-02	3.35e-04	3.42e-03
Si-32	2.09e-16	2.16e-14	1.74e-14	Cd-109	9.07e-09	8.36e-08	7.29e-07
Cl-36	1.38e-11	1.50e-10	1.57e-09	Cd-113m	2.45e-08	6.86e-05	5.48e-04
Ar-39	3.33e-14	0.00e+00	0.00e+00	In-115	2.30e-21	2.53e-17	8.10e-17
Ar-42	Not in FGR 11 or 12			Sn-119m	4.15e-07	1.02e-06	1.73e-05
K-40	2.73e-15	3.85e-17	4.39e-15	Sn-121m	2.57e-10	1.72e-09	2.47e-08
Ca-41	0.00e+00	1.49e-13	1.07e-11	Sn-126	5.11e-06	5.19e-08	8.19e-07
V-49	0.00e+00	0.00e+00	0.00e+00	Sb-125	1.94e-02	1.31e-04	2.61e-03
V-50	Not in FGR 11 or 12			Te-123	1.20e-20	2.28e-20	6.86e-19
Mn-54	4.64e-06	7.15e-09	2.24e-07	I-129	2.15e-10	3.42e-09	4.13e-07
Fe-55	0.00e+00	1.63e-08	2.79e-07	Cs-134	1.00e+00	5.79e-03	6.96e-01
Co-60	8.77e-04	1.40e-05	1.31e-04	Cs-135	6.88e-12	9.70e-10	1.14e-07
Ni-59	0.00e+00	1.66e-11	9.79e-11	Cs-137	1.81e-01	2.01e-03	2.38e-01
Ni-63	0.00e+00	6.27e-09	4.36e-08	Ba-133	1.75e-36	8.16e-39	2.70e-37
Zn-65	2.44e-04	1.59e-06	8.55e-05	La-137	0.00e+00	0.00e+00	0.00e+00
Se-79	3.75e-12	2.35e-09	1.58e-07	La-138	7.05e-15	1.44e-15	4.69e-16
Kr-81	1.05e-14	0.00e+00	0.00e+00	Ce-142	Not in FGR 11 or 12		
Kr-85	6.17e-05	0.00e+00	0.00e+00	Ce-144	1.71e-01	2.33e-01	1.00e+00
Rb-87	1.37e-15	3.73e-14	4.31e-12	Nd-144	Not in FGR 11 or 12		
Sr-90	8.11e-04	5.09e-02	4.53e-01	Pm-145	0.00e+00	0.00e+00	0.00e+00
Zr-93	0.00e+00	3.24e-07	1.27e-07	Pm-147	2.27e-06	2.11e-03	4.28e-03
Nb-91	Not in FGR 11 or 12			Pm-146	8.39e-06	3.31e-07	6.28e-07
Nb-93m	6.54e-12	2.18e-09	2.95e-09	Sm-145	0.00e+00	0.00e+00	0.00e+00
Nb-94	8.24e-10	4.18e-11	5.47e-11	Sm-146	0.00e+00	1.10e-11	2.05e-12
Mo-93	2.15e-13	1.23e-11	4.42e-11	Sm-147	0.00e+00	4.46e-11	8.40e-12
Tc-97	0.00e+00	0.00e+00	0.00e+00	Sm-148	Not in FGR 11 or 12		
Tc-98	3.48e-11	1.10e-13	1.78e-12	Sm-149	Not in FGR 11 or 12		
Tc-99	7.79e-10	6.13e-08	8.16e-07	Sm-151	1.72e-10	6.23e-06	6.13e-06
Ru-106	4.30e-01	1.88e-01	8.20e-01	Eu-152	1.68e-05	6.26e-07	1.39e-06

Table D-5 (continued)

Nuclide	Ground	Inhalation	Ingestion	Nuclide	Ground	Inhalation	Ingestion
Eu-154	5.38e-02	2.37e-03	6.01e-03	U-233	7.03e-15	8.08e-10	1.31e-10
Eu-155	8.27e-04	2.23e-04	6.24e-04	U-234	1.32e-10	5.17e-05	8.40e-06
Eu-150	7.03e-11	2.56e-12	4.62e-12	U-235	2.89e-09	5.57e-07	9.20e-08
Gd-152	0.00e+00	2.76e-17	1.38e-18	U-236	2.16e-11	1.50e-05	2.43e-06
Gd-153	6.08e-06	7.01e-07	2.62e-06	U-238	1.80e-08	1.67e-05	2.87e-06
Tb-157	0.00e+00	0.00e+00	0.00e+00	Np-235	1.42e-11	2.16e-11	9.59e-11
Ho-163	Not in FGR 11 or 12			Np-236	1.71e-12	4.58e-10	2.89e-10
Ho-166m	3.24e-08	2.88e-09	2.28e-09	Np-237	1.76e-07	1.03e-04	6.40e-05
Tm-171	2.01e-12	1.95e-11	6.96e-11	Pu-236	1.10e-10	8.24e-05	5.04e-05
Lu-176	4.83e-33	1.50e-33	1.26e-33	Pu-238	2.51e-07	7.71e-01	4.77e-01
Hf-182	0.00e+00	0.00e+00	0.00e+00	Pu-239	3.99e-08	6.88e-02	4.30e-02
Ta-180	0.00e+00	0.00e+00	0.00e+00	Pu-240	3.36e-08	1.17e-01	7.30e-02
Re-187	0.00e+00	1.81e-19	2.41e-18	Pu-241	1.35e-06	7.00e-01	4.41e-01
Os-194	5.32e-17	7.74e-17	1.41e-16	Pu-242	1.74e-10	6.63e-04	4.12e-04
Ir-192m	1.84e-14	1.68e-15	2.25e-15	Pu-244	1.38e-12	3.28e-10	2.05e-10
Pt-190	Not in FGR 11 or 12			Am-241	2.83e-06	3.41e-02	2.12e-02
Pt-193	1.73e-19	8.25e-18	3.27e-16	Am-242m	2.73e-07	2.09e-03	1.30e-03
Tl-204	0.00e+00	0.00e+00	0.00e+00	Am-243	1.68e-05	9.82e-03	6.14e-03
Pb-204	Not in FGR 11 or 12.			Cm-243	1.14e-05	7.12e-03	4.42e-03
Pb-205	6.92e-21	4.56e-18	1.44e-16	Cm-244	4.28e-07	1.00e+00	6.17e-01
Pb-210	1.39e-17	6.26e-14	1.49e-12	Cm-245	1.22e-07	1.93e-04	1.20e-04
Bi-208	Not in FGR 11 or 12			Cm-246	1.24e-11	5.71e-05	3.55e-05
Bi-210m	1.31e-14	8.51e-14	8.16e-14	Cm-247	8.17e-13	2.16e-10	1.35e-10
Ra-226	7.07e-14	6.43e-14	7.53e-13	Cm-248	1.31e-16	2.92e-09	1.82e-09
Ra-228	5.70e-18	5.73e-18	1.24e-16	Cm-250	4.83e-19	2.83e-15	1.77e-15
Ac-227	3.12e-13	1.24e-09	2.06e-10	Bk-249	4.75e-14	1.16e-08	7.59e-09
Th-228	1.26e-08	5.06e-07	8.98e-08	Cf-249	4.21e-12	1.56e-09	9.69e-10
Th-229	1.46e-13	2.35e-10	3.32e-11	Cf-250	1.27e-14	3.34e-08	2.06e-08
Th-230	9.83e-15	3.14e-09	4.01e-10	Cf-251	3.71e-13	4.91e-10	3.07e-10
Th-232	4.79e-21	1.79e-14	2.26e-15	Cf-252	3.21e-14	3.39e-08	1.78e-08
Pa-231	1.06e-12	8.47e-09	5.30e-09	Es-254	1.11e-12	9.71e-12	5.64e-12
U-232	5.60e-12	4.85e-06	7.31e-07				

RECOMMENDED RADIONUCLIDES FOR INCLUSION

Table D-6 lists all radionuclides with half-lives greater than six months which were included in the present review. A check mark (✓) in the right-most ("USE") column of Table D-6 indicates that that radionuclide is recommended for inclusion in the scrap recycle analysis. The basis for these recommendations is discussed below.

Basis for Recommendations. A recommendation to include a radionuclide in the scrap recycle analysis is based on the following:

- Each of the sources reviewed was assigned a weighting factor, depending on its applicability to scrap recycle. The weighting factors range from 6 for those sources which are most applicable to scrap recycle to 2 for those documents which are least applicable. These weighting factors are shown in parentheses below the designation of each source document in the heading of Table D-6.
- For each radionuclide identified in one or more of the sources reviewed, a score was calculated by simply adding the weighting factors for each source in which the radionuclide appeared. These scores are shown in the second column from the right on Table D-6.
- Those radionuclides with a score of 10 or greater are recommended for inclusion in the scrap recycle analysis, as indicated by a check mark in the right-most column of Table D-6.
- Members of the thorium and uranium radioactive decay series have been recommended for inclusion even if they have scores below 10, to allow these series to be analyzed.

Table D-6 (continued)

Nuclide	Source (weighting factor)									Score	Use
	NUREG/ CR-0134 (5)	IAEA 95 (6)	WINCO 1191 (4)	NUREG/ CR-0130 (4)	NUREG/ CR-3585 (3)	NUREG/ CR-4370 (2)	SAND 92-0700 (2)	ORIGEN (2)	SAND 91-2795 (2)		
Cd-113m	--	--	--	--	--	--	--	✓	--	2	--
Sn-121	--	--	--	--	--	--	--	--	✓	2	--
Sn-126	--	--	--	--	✓	✓	--	--	✓	7	--
Sb-125	--	--	✓	--	✓	✓	✓	✓	--	13	✓
I-129	--	--	✓	--	✓	✓	--	--	✓	11	✓
Cs-134	✓	✓	--	✓	✓	✓	✓	✓	--	24	✓
Cs-135	--	--	--	--	✓	✓	--	--	✓	7	--
Cs-137	✓	✓	--	✓	✓	✓	✓	✓	✓	26	✓
Ce-144	✓	✓	✓	--	✓	--	✓	✓	--	22	✓
Pm-147	--	✓	--	--	--	--	✓	✓	--	10	✓
Sm-151	--	--	--	--	--	--	--	--	✓	2	--
Eu-152	--	✓	--	--	✓	✓	✓	--	--	13	✓
Eu-154	--	--	--	--	✓	✓	✓	✓	--	9	--
Eu-155	--	--	--	--	--	--	✓	✓	--	4	--
Pb-210	--	--	--	--	✓	--	--	--	✓	5	✓
Ra-226	--	--	--	--	✓	--	--	--	✓	5	✓
Ra-228	--	--	--	--	✓	--	--	--	--	3	✓
Ac-227	--	--	--	--	✓	--	--	--	✓	5	✓
Th-228	--	--	--	--	✓	--	--	--	--	3	✓
Th-229	--	--	--	--	✓	--	--	--	✓	5	✓
Th-230	--	--	--	--	✓	--	--	--	✓	5	✓
Th-232	--	--	--	--	✓	--	✓	--	--	5	✓
Pa-231	--	--	--	--	✓	--	--	--	✓	5	✓
U-232	--	--	--	--	✓	--	--	--	✓	5	--

Table D-6: Selection of Nuclides to Be Included in Scrap Recycle Analysis

Nuclide	Source (weighting factor)									Score	Use
	NUREG/ CR-0134 (5)	IAEA 95 (6)	WINCO 1191 (4)	NUREG/ CR-0130 (4)	NUREG/ CR-3585 (3)	NUREG/ CR-4370 (2)	SAND 92-0700 (2)	ORIGEN (2)	SAND 91-2795 (2)		
H-3	--	--	--	--	✓	✓	--	--	--	5	--
C-14	✓	--	✓	--	✓	✓	--	--	✓	16	✓
Na-22	✓	--	--	--	✓	--	--	--	--	8	--
Cl-36	--	--	--	--	✓	--	--	--	✓	5	--
Mn-54	✓	✓	✓	✓	✓	--	✓	--	--	24	✓
Fe-55	✓	✓	✓	✓	✓	✓	--	--	--	24	✓
Co-57	--	--	✓	--	✓	--	--	--	--	7	--
Co-60	✓	✓	✓	✓	✓	✓	✓	✓	--	28	✓
Ni-59	✓	--	✓	✓	✓	✓	--	--	✓	20	✓
Ni-63	✓	✓	✓	✓	✓	✓	✓	--	✓	28	✓
Zn-65	✓	✓	✓	✓	✓	--	--	✓	--	24	✓
Se-79	--	--	--	--	--	--	--	--	✓	2	--
Rb-86	--	--	--	--	✓	--	--	--	--	3	--
Sr-90	✓	✓	--	✓	✓	✓	✓	✓	✓	26	✓
Zr-93	--	--	--	--	--	--	--	--	✓	2	--
Nb-93m	--	--	✓	--	--	--	--	--	--	4	--
Nb-94	--	✓	✓	✓	✓	✓	--	--	✓	21	✓
Mo-93	--	--	✓	✓	--	--	--	--	✓	10	✓
Tc-99	✓	✓	--	--	✓	✓	✓	--	✓	20	✓
Ru-106	✓	✓	--	✓	✓	✓	✓	✓	--	24	✓
Pd-107	--	--	--	--	--	--	--	--	✓	2	--
Ag-108m	--	--	--	--	✓	--	--	--	✓	5	--
Ag-110m	--	✓	✓	--	✓	--	--	✓	--	15	✓
Cd-109	--	--	--	--	✓	--	--	--	--	3	--

Table D-6 (continued)

Nuclide	Source (weighting factor)									Score	Use
	NUREG/ CR-0134 (5)	IAEA 95 (6)	WINCO 1191 (4)	NUREG/ CR-0130 (4)	NUREG/ CR-3585 (3)	NUREG/ CR-4370 (2)	SAND 92-0700 (2)	ORIGEN (2)	SAND 91-2795 (2)		
U-233	--	--	--	--	✓	--	✓	--	✓	7	--
U-234	✓	✓	--	--	✓	✓	--	--	✓	18	✓
U-235	✓	✓	--	--	✓	✓	✓	--	✓	20	✓
U-236	--	--	--	--	✓	✓	✓	--	✓	9	--
U-238	✓	✓	--	--	✓	✓	✓	--	✓	20	✓
Np-237	--	✓	--	--	✓	✓	✓	✓	✓	17	✓
Pu-236	--	--	--	--	✓	✓	--	--	--	5	--
Pu-238	--	--	✓	--	✓	✓	✓	✓	✓	15	✓
Pu-239	✓	✓	✓	--	✓	✓	✓	✓	✓	26	✓
Pu-240	--	✓	✓	--	✓	✓	✓	✓	✓	21	✓
Pu-241	--	✓	--	--	✓	✓	✓	✓	✓	17	✓
Pu-242	--	--	--	--	✓	✓	✓	✓	✓	11	✓
Pu-244	--	--	--	--	✓	--	--	--	--	3	--
Am-241	✓	✓	--	--	✓	✓	✓	✓	✓	22	✓
Am-242	--	--	--	--	--	--	--	--	✓	2	--
Am-242m	--	--	--	--	--	--	--	✓	--	2	--
Am-243	--	--	--	--	✓	✓	--	✓	✓	9	--
Cm-242	--	--	--	--	--	✓	--	--	--	2	--
Cm-243	--	--	--	--	✓	✓	--	✓	✓	9	--
Cm-244	--	✓	✓	--	✓	✓	✓	✓	✓	21	✓
Cm-245	--	--	--	--	--	--	--	✓	✓	4	--
Cm-246	--	--	--	--	--	--	--	--	✓	2	--
Cm-248	--	--	--	--	✓	--	--	--	--	3	--
Cf-252	--	--	--	--	✓	--	✓	--	--	5	--

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APPENDIX E

DISTRIBUTION OF RADIONUCLIDES DURING MELTING OF CARBON STEEL

Contents

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Tables

DISTRIBUTION OF RADIONUCLIDES DURING MELTING OF CARBON STEEL

E.1 INTRODUCTION

During the melting of potentially contaminated steel, the contaminants may be distributed among the metal product, the home scrap, the slag, the furnace lining and the off-gas collection system. In addition, some contaminants could pass through the furnace system and be vented to the atmosphere. In order to estimate the radiological impacts of recycling potentially contaminated scrap steel, it is essential to understand how the contaminants are distributed within the furnace system.

For example, a gaseous chemical element (*e.g.*, radon) will be exhausted directly from the furnace system into the atmosphere while a relatively non-volatile element (*e.g.*, manganese) can be distributed among all the other possible media. This distribution of potential contaminants is a complex process that can be influenced by numerous chemical and physical factors, including composition of the steel bath, chemistry of the slag, vapor pressure of the particular element of interest, solubility of the element in molten iron, density of the oxide(s), steel melting temperature and melting practice (*e.g.*, furnace type and size, melting time, method of carbon adjustment and method of alloy additions).

This appendix discusses the distribution of various elements with particular reference to electric arc furnace (EAF) steelmaking. The next three sections consider the calculation of partition ratios for elements between metal and slag based on thermodynamic considerations.[1] Section E.5 presents laboratory and production measurements of the distribution of various elements among slag, metal and the off-gas collection system. Section E.6 proposes distributions for those elements where theoretical or practical information is lacking, and Section E.7 provides recommendations for the assumed distribution of each element of interest.

E.2 THERMODYNAMIC CALCULATION OF PARTITION RATIOS

Partitioning of a solute element between a melt and its slag under equilibrium conditions can be calculated from thermodynamic principles if appropriate data are available. Consider a

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[1] Reference to a given element does not necessarily imply that it is in the elemental form. For instance, a metallic element might be found in the elemental state in the melt while its oxide is found in the slag.

divalent solute element M, such as cobalt, dissolved in molten iron, which reacts with iron oxide in the slag according to the following equation:

$$\underline{M} + FeO_{(slag)} = MO_{(slag)} + Fe_{(l)} \qquad (E\text{-}1)$$

where $\underline{M}$ is the symbol for solute dissolved in liquid iron.

Equation E-1 can be written as the difference between the following equations:

$$\underline{M} + \frac{1}{2}O_2 = MO \qquad (E\text{-}2)$$

and

$$Fe + \frac{1}{2}O_2 = FeO \qquad (E\text{-}3)$$

The Gibb's free energy for Equation E-1, ΔF°_1 can be expressed as the difference in the free energies of Equations E- 2 and E- 3, *viz.*:

$$\Delta F^{\circ}_1 = \Delta F^{\circ}_2 - \Delta F^{\circ}_3$$

Thermodynamic data for Equation E-2 are normally tabulated assuming that the standard state for M is the pure liquid or solid, but it is often desirable to convert from the pure elemental standard state to a hypothetical standard state where M is in a dilute solution. In steelmaking, 1 wt% M in solution in iron is commonly used for this new standard state[2] as defined by the transformation:

$$M_{(pure)} = \underline{M} \qquad (E\text{-}4)$$

The free energy change for M from the pure state to $\underline{M}$ in the dilute state is (DAR53):

$$\Delta F^{\circ}_4 = RT \ln\left(\frac{\gamma^{\circ} M_{Fe}}{100\, M_M}\right)$$

where:

---

[2] Concentrations are expressed here as wt% instead of mass % since wt% is commonly used in the steelmaking literature. The terms are synonymous.

T = absolute temperature in kelvin (K)

R = universal gas constant
= 1.987 cal/mole·K

$\gamma°_M$ = Henry's Law activity[3] coefficient (based on atom fraction) of M at infinite dilution in iron

M_{Fe} = atomic weight of iron
= 55.85

M_M = atomic weight of M

Equation E-2 can also be written as the difference of Equation E-5 (below) and Equation E-4.

$$M_{(pure)} + \tfrac{1}{2}O_2 = MO \qquad \text{(E-5)}$$

Therefore, $\Delta F°_2 = \Delta F°_5 - \Delta F°_4$ and the Gibb's free energy change for Equation E-1 can be written as

$$\begin{aligned}\Delta F_1^\circ &= \Delta F_5^\circ - \Delta F_3^\circ - \Delta F_4^\circ \\ &= \Delta F^\circ_{f,MO} - \Delta F^\circ_{f,FeO} - RT\ln\left(\frac{\gamma^\circ M_{Fe}}{100\,M_M}\right)\end{aligned} \qquad (6)$$

where $\Delta F°_f$ is the free energy of formation of the particular oxide.

At equilibrium

$$\begin{aligned}\Delta F_1^\circ &= -RT\ln K_1 \\ &= -RT\ln\left(\frac{a_{Fe}\,a_{MO}}{a_{FeO}\,a_M}\right)\end{aligned} \qquad (7)$$

where a is the activity of each species in Equation E-1 and K_1 is the equilibrium constant. In the steel bath, a_{Fe} can be assumed to be 1, while $a_{FeO} = \gamma_{FeO}N_{FeO}$. To estimate N_{FeO}, the mole fraction of FeO in the slag, the nominal composition of the slag was assumed to be 50 wt% CaO, 30 wt% SiO_2 and 20 wt% FeO. Thus, $N_{FeO} = 0.167$. Various investigators have described the activity of

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[3] In Sections E.1, E.2 and E.3, activity refers to thermodynamic activity, not radioactivity.

FeO in ternary mixtures of CaO, FeO and SiO_2 (PHI51, ANS84). For the slag composition assumed here, based on the ternary diagram in ANS84, when N_{FeO} is 0.2, a_{FeO} is about 0.4 (*i.e.*, γ_{FeO} is about 2). Consequently, $a_{FeO} = 0.333$.

For the dilute standard state, $a_{\underline{M}}$ is equal to wt% $\underline{M}$ and, for dilute solutions of MO in the slag, one can assume that $a_{MO} = N_{MO}$. It follows that

$$\frac{N_{MO}}{\text{wt\% } \underline{M}} = a_{FeO} \exp\left(\frac{-\Delta F_i^\circ}{RT}\right) \tag{8}$$

where $\frac{N_{MO}}{\text{wt\% } \underline{M}}$ is one form of the partition ratio for M between the melt and the slag.

For metal oxides other than those formed from divalent cations, the different stoichiometries must be accommodated in Equations E-6, E-7 and E-8.

Using values of γ° for various solute elements in iron at 1,873 K tabulated by Sigworth and Elliott (SIG74)[4] and free energy of formation data for oxides tabulated by Glassner (GLA57), partition ratios between melt and slag were calculated for this report and are presented in Table E-1. Values in the last column of Table E-1 will be described in Section E.3.

When the partition ratio is large, the solute element is strongly concentrated in the slag under equilibrium conditions. This is true for Al, Ce, Nb, Ti, U and Zr, which all have partition ratios (as defined here) of 80,000 or greater. Similarly, when the partition ratio is small, the solute element is concentrated in the molten iron. Examples of this are Ag, Co, Cr, Cu, Ni, Pb, Sn, Mo and W, which all have partition ratios of 0.008 or less. Mn, Si and V, with partition ratios ranging from about 3 to 40, are expected to be more evenly distributed between melt and slag. Ag will not react with FeO in the slag, so on the basis of slag/metal equilibria, this element should remain in the melt. However, Ag has a relatively high vapor pressure at steelmaking temperatures (*i.e.*, 10^{-2} atm at 1,816 K), so some would tend to be removed at a rate dependent on the rate of transfer of Ag vapor through the slag.

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[4] The value of γ° for Ce is from ANS84. A compendium of values for γ° similar to that in SIG74 has been prepared by the Japan Society for the Promotion of Science (JAP88). Some differences exist between values in SIG74 and JAP88, particularly for W, Co, Pb and Ti. JAP88 proposes a value of γ° for $Ce_{(l)}$ of 0.332. This difference in γ° values does not affect the conclusions about Ce partitioning.

Table E-1. Partition Ratios at 1,873 K for Various Elements Dissolved in Iron and Slag

M	Oxide	γ°_{M}	ΔF°_{fMO} (kcal/mole)[a]	Partition Ratio	
				(N_{MO}/wt%M)	(mass in slag/ mass in metal)
$Ag_{(l)}$	Ag_2O	200	+20.6	3.89E-04[b,c]	
$Al_{(l)}$	Al_2O_3	0.029[d]	-257	1.32E+05[b]	
$Ca_{(g)}$	CaO	2240	-104	1.53E+09	1.1E+10
$Ce_{(l)}$	CeO_2	0.026	-176	4.33E+07	1.1E+09
$Co_{(l)}$	CoO	1.07	-18.2	4.79E-05	5.0E-04
$Cr_{(s)}$	Cr_2O_3	1.14	-80.0	1.21E-04[b]	
$Cu_{(l)}$	Cu_2O	8.6	-11.0	1.99E-03[b]	
$Mn_{(l)}$	MnO	1.3[e]	-58.0	2.74E+00	2.7E+01
$Mo_{(s)}$	MoO_3	1.86	-89.1	1.23E-05	2.1E-4
$Nb_{(s)}$	Nb_2O_5	1.4	-275	8.12E+04[b]	
$Ni_{(l)}$	NiO	0.66	-19.0	3.72E-05	3.9E-04
$Pb_{(l)}$	PbO	1400	-15.5	8.55E-03	3.2E-01
$Si_{(l)}$	SiO_2	0.0013	-129	3.76E+01	1.9E+02
Sn	SnO_2	2.8	-47.6	6.07E-06	1.3E-04
$Ti_{(s)}$	TiO_2	0.038	-147	7.72E+04	6.6E+05
$U_{(l)}$	UO_2	0.027	-180	8.87E+07	3.8E+09
$V_{(s)}$	V_2O_5	0.1	-206	7.68E+00[b]	
$W_{(s)}$	WO_3	1.2	-96.2	2.77E-05	9.1E-04
$Zr_{(s)}$	ZrO_2	0.037	-178	1.59E+08	2.6E+09

[a] $\Delta F^{\circ}_{fFeO} = -34.0$ kcal/mole

[b] $PR = N^{1/2}$/wt% $\underline{M}$

[c] Ag will not react with FeO, Ag_2O unstable at 1,873K

[d] According to ANS84, $\gamma^{\circ}_{Al} = 0.005$

[e] According to ANS84, $\gamma^{\circ}_{Mn} = 1.48$

It is instructive to examine the impact of assuming a dilute solution in iron rather than the pure element as the standard state for the solute. For those elements that tend to partition strongly to the melt (Co, Cr, Cu, Mo, Ni, Sn and W), change of standard state from the pure metal to the dilute solution increases partitioning to the melt by factors of about 10 to 300. Lead is an exception, presumably due to its strong deviation from ideal solution behavior. Similarly, use of a dilute solution as the standard state decreases partitioning to the slag for the strong oxide formers such as Al, Ce, Nb, Ti, U and Zr by factors of about 100 to 16,000. The exception is calcium with strong positive deviation from ideality. These observations emphasize the importance of using a dilute solution as the standard state when adequate data are available.

As noted previously, the calculations in Table E-1 assumed, for simplicity, that the activity of MO in the slag was equal to the mole fraction (*i.e.*, $\gamma_{MO} = 1$). This may not be a good assumption. If, for example, $\gamma_{MO} = 0.01$, N_{MO} would increase 100-fold. Work by Ostrovski on the partitioning of tungsten in steel melted in a 25-ton electric arc furnace illustrates the impact of melting practice and slag chemistry on the activity of WO_3 in the slag (OST94). When the steel was melted under strongly oxidizing conditions utilizing a 30-minute oxygen blow, the activity coefficient was found to be a function of the ratio %CaO:%SiO_2 in the slag and varied from about 10^{-2} to about 10^{-4} as the CaO:SiO_2 ratio increased from 1:1 to 4:1. Typical measured values of $\log \frac{(wt\%\ W)}{[wt\%\ W]}$ were between 1 and 2 where (% W) and [% W] are the tungsten contents of the slag and the metal, respectively.[5] A good fit between experimental and calculated partition ratios was obtained using the following equations:

$$\log \gamma_{WO_3} = -2.076 - 0.592 \frac{(\%CaO)}{(\%SiO_2)}$$

and

$$\log \frac{(\%W)}{[\%W]} = \frac{3054}{T} - 4.56 - \log \gamma_{WO_3} + 3 \log a_{FeO} + \log \frac{M_W}{M_{WO_3}} + \log\left[M_{WO_3}\left(n_{FeO} + n_{CaO} + n_{SiO_2} + n_{WO_3}\right)\right]$$

where n is the number of moles per 100 grams of the various slag components. With this melting practice, approximately 94% of the tungsten in the feed was transferred to the slag, 4% remained in the melt and the balance was lost. This emphasizes that special melting practices can produce substantially different results from the predictions in Table E-1.

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[5] The convention of using (x) and [y] to signify concentrations or components in the slag and the metal, respectively, is commonly used in the technical literature and will generally be used in this report.

The thermodynamic treatment used to derive the partition ratios in Table E-1 assumes that the melt is a binary system of iron and solute M, while in practice the melt will actually be a multi-component solution. In recent years, a considerable amount of work has been done to develop, both theoretically and experimentally, a solution model which considers interactions between solute elements (ENG92, SIG74, ANS84). The activity of element *i* in dilute solution can be expressed as:

$$a_i = f_i \text{ (wt\% i)}$$

where f_i is the Henry's Law activity coefficient (for concentrations expressed in wt%). The first order interaction coefficients e_i^j are defined by the equation

$$\log f_i = \sum_{j=2}^{k} e_i^j \text{ (\% j)}$$

(Higher order terms are possible but are not considered here.) Using, for illustrative purposes, a low alloy 4140 steel with the nominal composition 0.4% C, 0.04% S, 0.9% Cr and 0.1% Co, and the interaction coefficients for cobalt with these elements in liquid iron from ENG92, f_{Co} was calculated to be 0.975. For this example, the impact of the binary interactions on Co activity in iron is quite small. Unfortunately, interaction coefficients for many of the elements of interest in the melting of potentially contaminated scrap metals are not available to refine the calculations summarized in Table E-1.

E.3 CORRELATION WITH OTHER FORMS OF PARTITION RATIO

In the literature, the partition ratio (PR) may be expressed in a variety of ways. For example, in Chapter 9 of SCA95, partition ratios are expressed as "mass in slag/mass in steel." It is of interest to compare this formulation with the definition in column 5 of Table E-1 (*i.e.*, N_{MO}/wt%$\underline{M}$). The SCA95 PR may be expanded as:

$$PR = \frac{(\text{wt\% M})\, m_g}{[\text{wt\% M}]\, m_s} \qquad (9)$$

where:

m_g = mass of slag
m_s = mass of steel

and, if one assumes that the relevant reaction is that in Equation E-2 above, one can write:

$$PR = \frac{(\text{wt\% MO})\, m_s M_M}{[\text{wt\% }\underline{\text{M}}]\, m_s M_{MO}} \tag{10}$$

where M_M and M_{MO} are the atomic weight of M and the molecular weight of MO, respectively.

Equation E-10 is based on the premise that the reaction involves a divalent solute metal. It is equally true for all oxides where the ratio of the anion to the cation is a whole number. For simplicity, if one assumes that the slag consists of two oxide components MO and RO and that wt% MO is << wt% RO, then one can write that

$$N_{MO} = \frac{(\text{wt\% MO})/M_{MO}}{100/M_{RO}} \tag{11}$$

or that

$$(\text{wt\% MO}) = \frac{100\, N_{MO} M_{MO}}{M_{RO}} \tag{12}$$

which can be substituted into Equation E-10 to give

$$PR = \frac{100\, N_{MO} \cdot m_s M_M}{[\text{wt\% }\underline{\text{M}}]\, m_s M_{RO}} \tag{13}$$

Equation E-13 relates the partition ratio as defined in SCA95 to that in Table E-1. Column 6 of Table E-1 converts the partition ratios in column 5 to the formulation in SCA95 (*i.e.*, mass in slag/mass in metal), using the assumptions and simplifications described above, and further assuming that the ratio, mass of slag : mass of metal is 1:10 and RO is CaO. This conversion is only done for those oxides where the anion/cation ratio is a whole number.

E.4 ESTIMATES OF THE PARTITIONING OF OTHER CONTAMINANTS

Values of the Henry's Law activity coefficient (γ°_M) are not available for many solute elements of interest in recycling potentially contaminated steel scrap. However, an indication of partitioning between the melt and the slag can be obtained by calculating the Gibb's free energy for the reaction

$$M + \left(\frac{y}{x}\right)FeO = \left(\frac{y}{x}\right)Fe + \left(\frac{1}{x}\right)M_xO_y \qquad (14)$$

where M is the pure component rather than the solute dissolved in the melt and FeO and M_xO_y are slag components. Values of the standard free energy change for Equation E-14 are summarized in Table E-2 for all instances where the reaction occurs as written.

Table E-2 shows that Ac, Am, Ba, Np, Pa, Pu, Ra, Sm, Sr, Th and Y all will react with FeO to form their respective oxides as indicated by the calculated free energies. Thus, these elements should be preferentially distributed to the slag. By chemical analogy to similar species in Table E-1, one can estimate that the partition ratios (N_{MO}/wt% M) should be on the order of 10^4 or greater.[6] The solute elements Bi, Cd, Cs, Ir, K, Na, Re, Ru, Sb, Se, Tc and Zn do not react with FeO either because the oxides are unstable or because Equation E-14 is thermodynamically unfavorable. Of these elements, Ir, Re, Ru and Tc are expected to remain in the melt. As indicated in Table E-3, the solute elements Bi, Cd, Cs, Po, Sb, Se and Zn have low boiling points and would be expected to vaporize from the melt to some degree at typical steelmaking temperatures of 1,823 K to 1,923 K. For example, Cs would tend to be removed at a rate dependent on the rate of transfer of vapor through the slag unless some stable compound such as Cs_2SiO_3 forms in the slag. Should Cs_2O form during the melting process before a continuous slag had formed, it would be volatilized since the boiling point of the oxide is about 915 K. The boiling point of metallic cesium is in the same temperature range. Even though an element may have a low boiling point, it cannot be assumed, *a priori*, that the element will completely vaporize from the melt. Some may remain in the melt and some may be contained in the slag. For example, elements such as Ca, Mg, K and Na are found as oxides and silicates in steel slags (HAR90).

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[6] The free energies in Table E-2 were recalculated assuming that γ° in Equation E-6 was unity, and partition ratios were then calculated using Equation E-8. All partition ratios calculated in this manner for elements expected to partition to the slag were greater than 10^4 except Ba (6.3×10^3) and Ra (320). If all these calculated partition ratios were reduced by a factor of 10^3 to adjust for the fact that values of γ° are expected to be less than unity, estimated partition ratios are greater than 10^3 for all slag formers except Ba (6.3), Ra (0.321), and Sr (15). These three elements are in Group II of the periodic table and have electronic structures and chemical properties similar to Ca. As discussed previously in Section E.2, Ca has a value of $\gamma^\circ = 2{,}240$. By analogy, one would expect that the partition ratios of Ba, Ra, and Sr would actually be higher than calculated with $\gamma^\circ = 1$. For example, if $\gamma_{Ra}^\circ = 2{,}000$, the partition ratio for Ra, as defined by Equation E-8, would be 6×10^5.

For example, elements such as Ca, Mg, K and Na are found as oxides and silicates in steel slags (HAR90).

Pehlke has shown that, for a solute M dissolved in a solvent (liquid Fe), the following equation applies (PEH73):

$$P_M(T) = P_M{}^{\circ}(T)\ \gamma_M(T)\ N_M \tag{15}$$

where:

P_M = vapor pressure of M over melt
$P_M{}^{\circ}$ = vapor pressure of pure M
γ_M = activity coefficient of M in melt
N_M = mole fraction of M in melt

Thus, as the temperature of the melt increases, the quantity of the volatile element M in the melt decreases by an amount determined by the temperature dependency of $P_M{}^{\circ}$. Based on vapor pressure data for Pb, Sb and Bi from BRA92 and Zn from PER92, one can estimate that increasing the temperature of the iron bath from 1,873 K to 1,923 K will reduce the amount of Pb, Sb, or Bi by about 25% while that of Zn will be reduced by about 18% (assuming that γ_M is independent of temperature over the same range and P_M is constant). Actually, γ_M is an increasing function of temperature for Sb (NAS93) and a decreasing function for Zn (PER92).

E.5 OBSERVED PARTITIONING

This section discusses available experimental and production information on the distribution of possible contaminant elements among melt, slag and the off-gas collection system in steelmaking. Several of the key references are abstracted in Appendix E-1 which describes test conditions and relevant results from selected publications. Since many of the references cited in this section discuss the distribution of multiple elements in a single test, it would be cumbersome to repeat all the experimental details here for each element. Table E-4 summarizes the references by contaminant element. Substantial additional information on these and other references can be found in WOR 93. Some additional perspective concerning the concentrations of impurities and alloying elements can be obtained by examining the composition of a typical low carbon steel (i.e SAE 1020) as shown below:

carbon 0.18–0.23%

manganese 0.60–0.90%
phosphorus .... ≤ 0.04%
sulfur .... ≤ 0.05%

Thus the steel melting process must control carbon and manganese within specified ranges and insure that the maximum concentrations of sulfur and phosphorus are not exceeded. The furnace charge, the melting conditions and the slagging practice must all be carefully managed to achieve the desired steel chemistry. .

E.5.1 Americium

Based on the thermodynamic equilibria, americium would be expected to partition strongly to the slag. Gomer of British Steel reported that, when melting reactor heat exchanger tubing contaminated with Am-241 in a 5-ton electric arc furnace, traces of Am-241 were found in the slag. No other Am-241 was detected (PFL85). In laboratory steel melting experiments in a 5-kg furnace, the Am-241 distribution was 1% in the ingot, 110%[7] in the slag and 0.05% in the aerosol off-gas filter, resulting in a partition ratio between slag and metal of about 100 (SCH90, SCH88). Americium is chemically similar to uranium which partitions strongly to the slag (HAR90). On the basis of the available information, Am is expected to partition to the slag as predicted by the thermodynamic calculations. However, one caveat is offered by Harvey (HAR90). Since the density of the AmO_2 is high (11.68 g/cm^3), transfer of Am to the slag may be retarded by gravity.

In small-scale laboratory experiments using mild steel (see Section E.5.20 for details), Am was observed to partition to the slag (GER77). Ratios of the concentration of Am in slag to the concentration of Am in metal generally exceed 1000:1.

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[7] Because of differences in detection efficiencies, more radioactivity is sometimes detected in the products than was measured in the furnace charge.

Table E-2. Standard Free Energy of Reaction of Various Contaminants with FeO at 1,873 K

Element	Oxide	ΔF° (kcal)	Comments
$Ac_{(l)}$	Ac_2O_3	-120	Ac should partition to slag
$Am_{(l)}$	Am_2O_3	-103	Am should partition to slag
$Ba_{(l)}$	BaO	-57.1	Ba should partition to slag
$Bi_{(g)}$	Bi_2O_3		Bi will not react with FeO, some may vaporize from melt
$Cd_{(g)}$	CdO		CdO unstable at 1873 K, Cd should vaporize from the melt
$Cs_{(l)}$	Cs_2O		Cs_2O unstable at 1873 K, Cs should vaporize from melt, some Cs may react with slag components
$Ir_{(s)}$	IrO_2		IrO_2 unstable above ≈1100 K, Ir should remain in melt
$K_{(g)}$	K_2O		K_2O less stable than FeO, other K compounds stable in slag
$Na_{(g)}$	Na_2O		Na_2O less stable than FeO, other Na compounds stable in slag
$Np_{(l)}$	NpO_2	-100	Np should partition to slag
$Pa_{(l)}$	PaO_2	-94.7	Pa should partition to slag
$Po_{(g)}$	PoO_2		PoO_2 unstable above ≈1300 K, Po assumed to vaporize from melt
$Pu_{(l)}$	Pu_2O_3	-87.6	Pu should partition to slag
$Ra_{(g)}$	RaO	-47.7	Ra should partition to slag
$Re_{(s)}$	ReO_2		Re will not react with FeO, Re should remain in melt
$Ru_{(s)}$	RuO_4		RuO_4 unstable above ≈1700 K, Ru should remain in melt
$Sb_{(g)}$	Sb_2O_3		Sb will not react with FeO, some may vaporize from melt
$Se_{(g)}$	SeO_2		Se will not react with FeO, some may vaporize from melt
$Sm_{(l)}$	Sm_2O_3	-102	Sm should partition to slag
$Sr_{(g)}$	SrO	-58.6	Sr should partition to slag, but low boiling point could cause some vaporization
$Tc_{(s)}$	TcO_2		Tc will not react with FeO, should remain in melt
$Th_{(s)}$	ThO_2	-142	Th should partition to slag
$Y_{(l)}$	Y_2O_3	-101	Y should partition to slag
$Zn_{(g)}$	ZnO		Zn will not react with FeO, Zn should vaporize from melt

Table E-3. Normal Boiling Point of Selected Potential Contaminants[a]

Contaminant	Normal Boiling Point (K)
Bi	1900
Cd	1038
Cs	963
Pb	2010
Po_2	1300
Ra	1410
S_2	1890
Se_2	1000
Sb_2	1890
Zn	1180

[a] From DAR53

E.5.2 Antimony

As described previously, antimony will not react with iron oxide in the slag and therefore is expected to remain in the melt. However, as noted in Table E-3, the normal boiling point of antimony (1890 K) is at steelmaking temperatures and at least some vaporization would be expected. Contrary to this prediction, British Steel reports "...that when antimony is added to steel it is recovered with high yield." (HAR90). This view is supported by Philbrook (PHI51) who observed that antimony is probably almost completely in solution in steel. On the other hand, Stubbles (STU84a) indicates that antimony is volatilized from scrap during EAF melting. In no case is adequate background information provided to support the statements.[8]

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[8] In a recent telephone conversation, Dr. J. R. Stubble, currently Manager of Technology at Charter Steel Company, advised that his conclusions in STU84a were based on the high vapor pressure of Sb rather than experimental steel melting evidence. He would not argue against Harvey's conclusions. (Private communication - July 1, 1996).

Table E-4

Selected References on the Distribution of Potential Contaminants During Steelmaking

Contaminant	References
Ag	SAP90, HAR90, MEN90
Am	PFL85, SCH90, SCH88
C	SCH90, STU84b
Ce	SAP90, HAR90
Co	NAK92, LAR85, PFL85, SAP90, LAR85a, SCH90, HAR90, SCH88, MEN90
Cr	STU84a
Cs	NAK92, LAR85a, LAR85b, PFL85, SAP90, HAR90, MEN90
Eu	SAP90, LAR85a, HAR90
Fe	SCH90, SCH88
H	STU84b
Ir	LAR85b
Mn	NAK92, SAP90, STU84a, MER93, HAR90, MEN90
Mo	STU84a, CHE93
Nb	STU84a, HAR90
Ni	HAR90, STU84a, SCH90
P	STU94b
Pb	STU84a
Pu	GER77, HAR90
Ra	STA61
S	STU84b
Sb	HAR90, MEN90, STU84a, KAL91, NAS93
Sr	NAK92, LAR85b, SCH90
Th	HAR90
U	HAR90, LAR85a, SCH90, HES81, ABE85
Zn	HAR90, NAK92, SAP90, STU84a, MEN90
Zr	STU84a

Kalcioglu and Lynch found that Sb could be removed from carbon-saturated iron (typical of blast furnace operations) if temperatures exceeded 1,823 K and the slag basicity was greater than 1 (KAL91). Using very small samples where the mass of the slag was two grams and the mass of steel was three grams, about 45% to 51% of the antimony was vaporized at 1,823 K when the slag basicity, as defined by the following expression, was unity.

$$B = \frac{(CaO) + (MgO)}{(SiO_2) + (Al_2O_3)} = 1$$

The balance was distributed between slag and metal as follows :

Distribution of Sb Between Slag and Metal

[wt%Sb]	L_{Sb}[a]
0.40	0.55
0.46	0.59
0.51	0.67

[a] L_{Sb} = (wt%Sb)/[wt%Sb]

When the slag basicity was 0.818, values of L_{Sb} ranged from 0.09 to 0.13, and when the basicity was 0.666, L_{Sb} ranged from 0.05 to 0.08 at 1,823 K. The reaction which caused the marked increase in Sb partitioning to the slag when the basicity was increased to 1 was not identified.

In a proposed follow-on study to the work of Kalcioglu and Lynch, Zhong suggested that the reaction

$$2\underline{Sb} + 3(FeO) + (O^{2-}) = 2(SbO_2^-) + 3Fe_{(l)}$$

has an estimated value for ΔF° of -4,000 cal (ZHO94). While not strongly favoring partition to the slag, the reaction can proceed as written particularly since a_{FeO} and a_{O2-} tend to be high in basic slags. Using data presented by Zhong, the partition ratio for the above reaction can be roughly estimated to be 0.006—a value similar to those for Cu and Pb in Table E-1.[9] The calculation supports the conclusion that Sb will not partition to the slag to a significant degree.

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[9] This calculation uses a value for γ°_{Sb} measured in carbon-saturated iron.

This conclusion is reinforced by the work of Nassaralla and Turkdogan (NAS93) who stated that "....most of the antimony will remain in the metal phase. However, it should be possible to remove some antimony from the hot metal by intermixing it with lime-rich flux under highly reducing conditions." Using values of γ°_{Sb} developed by these investigators, one can calculate a partition ratio for Sb of 8×10^{-6} at 1,873 K.

Based on calculated partition ratios (above and in Table E-1), vapor pressures of the pure metals (Table E-3) and vapor pressures of the metal oxides[10], one would expect that Sb and Pb would behave similarly. It is not clear why this is not the case since Sb tends to remain in the melt and Pb is primarily collected in the bag house. This may be a manifestation of significantly higher activity of Pb as compared to Sb in molten iron.

Menon *et al*. measured the distribution of Sb-125 from two heats of stainless steel (MEN90). Activities of 4.3×10^5 Bq were detected in the melt and 1.7×10^3 Bq in the baghouse dust. None was reported in the slag.

E.5.3 Carbon

Carbon is a carefully controlled element in steelmaking. Often excess carbon is added to the melt and reduced to its final level by oxygen decarburization. This process promotes slag/metal reactions and assists in removing hydrogen from the melt (STU84b). CO produced by the decarburization reaction combines with atmospheric oxygen in the off-gas to form CO_2, which is exhausted from the system (PHI51). If, for example, 10 lb of charge carbon per ton are added to a melt that nominally contains 5 lb of carbon per ton of scrap and the objective is to produce steel with a final carbon content of 0.2% (*i.e.*, an SAE 1020 steel), 0.55% C must be removed. Thus, about 73% of the carbon would be exhausted from the system and the balance would remain in the melt. The distribution of carbon between the melt and the off-gas is dependent upon the carbon content of the scrap charge, the melting practice (*i.e.*, use of charge carbon) and the desired carbon content of the finished steel.

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[10] Based on data from *Perry's Chemical Engineers Handbook* (6th ed.), the vapor pressures of PbO and Sb_4O_6 are one atmosphere at 1,745 K and 1,698 K, respectively.

E.5.4 Cerium

Based on thermodynamic calculations, Ce should strongly partition to the slag as CeO_2 or Ce_2O_3. Sappok has described experience in induction melting of contaminated steel from nuclear installations (SAP90). All Ce-144 contamination was found in the slag, although details of the melting and slagging practice were not discussed. Ce is sometimes added to steel to react with oxygen and sulfur. Since CeO_2 has a density of 6.9 g/cm^3, which is similar to that of molten steel, Harvey suggests that the density of the oxide retards transfer to the slag and, consequently, some CeO_2 may remain as non-metallic inclusions in the steel (HAR90).

According to JAP88, Ce_2O_3 rather than CeO_2 is the stable oxide during steelmaking. In addition, JAP88 recommends a value of 0.322 for γ° in dilute iron solutions. These differing assumptions do not alter the conclusion developed from the calculations in Section E.2 that Ce strongly partitions to the slag. Using the recommended data in JAP88, the partition ratio for Ce,

$\dfrac{N_{MO}^{\frac{1}{2}}}{\text{wt\% } \underline{M}}$, is 1.15 x 10^8.

E.5.5 Cesium

Based on free energy and vapor pressure considerations, Cs would be expected to volatilize from the melt. Furthermore, Cs has no solubility in liquid iron. According to ASM93:

> From the scant data reported here and by analogy with other iron-alkali metal binary phase diagrams, it is evident that Cs-Fe is virtually completely immiscible in the solid and liquid phases.

A number of investigators have reported measurements on the experimental distribution of Cs during steel melting. Sappok *et al.* observed that during air induction melting of about 2,000 tons of steel, no Cs-134/137 remained in the melt (SAP90). Cs was found both in the slag and in the dust collection system but the distribution was not quantified.

Nakamura and Fujuki of the Japanese Atomic Energy Research Institute (JAERI) obtained similar results from air induction melting of both ASTM-A335[11] and SUS 304 steels (NAK93).

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[11] This ASTM specification covers various seamless ferritic alloy steel pipes for high temperature service.

The Cs-137 was about equally distributed between the slag and the dust collection system, but only about 77% of the amount charged was recovered.

At the Idaho National Engineering Laboratory (INEL), Larsen *et al.* found Cs both in the slag and in the baghouse dust when melting contaminated scrap from the Special Power Excursion Reactor Test (SPERT) III (LAR85a). In tracer tests, Larsen *et al.* found that 5% to 10% of the Cs remained in Type 304L stainless steel ingots (LAR85b).

Gomer described results of three 5-ton electric arc furnace and one 500-kg induction furnace melts in which the chemical form of Cs addition and the slag chemistry were varied (PFL85, GOM85). Based on the fraction of Cs-134 recovered, the distribution of this nuclide is summarized in Table E-5, below.

Table E-5. Distribution of Cs-134 Following Steel Melting

Furnace Type	Cs Addition	Cs Distribution (%)			Cs Recovery (%)
		Steel	Slag	Off Gas	
EAF	CsCl	0	0	100	100
Induction	CsOH	0	100	0	91
EAF	CsOH	0	7	93	50
EAF	Cs_2SO_4	0	66	34	64

In the melt where the Cs addition was CsCl, the chloride, which is volatile below the steel melting temperature, was not collected in the slag because the slag had not formed before the CsCl had completely evaporated. In the induction furnace test, CsOH was added to the liquid steel under a quiescent acid slag. In the related arc furnace test with CsOH, the slag was not sufficiently acid to promote extensive formation of cesium silicate, which would be retained in the slag. In the arc furnace melt with the Cs_2SO_4 addition, this compound was apparently incorporated into the slag to a significant extent.

Harvey concluded that the hot, basic slags typical of EAF melting were not conducive to Cs retention in the slag (HAR90). A comparison of three arc furnace melts with varying slag compositions showed the following amounts of Cs retention in the slag 16 minutes after Cs was added to the melt:

- SiO_2:CaO = 3.1:1 .... 50% recovery
- SiO_2:CaO = 1.3:1 ..... < 4% recovery
- SiO_2:CaO = 0.41:1 ...... 0% recovery

In these tests, no Cs remained in the melt.

Menon *et al.* recounted that no Cs was found in the ingots or the slag after melting 332 tons of carbon steel in an induction furnace (MEN90), but that substantial Cs-137 (21,000 Bq/kg) was collected in the ventilation filters. During production of two heats of stainless steel, no Cs was found in the ingots; 32% was in the slag; and 68% in the baghouse dust (MEN90).

E.5.6 Chlorine

The disposition of chlorine depends on its form at the time of introduction into the EAF furnace. Any chlorine gas would be desorbed from the scrap metal surface and vented to the atmosphere. If the contaminant exists as a metal chloride, it is likely to be distributed between the slag and the baghouse dust. Cl^- has been reported in baghouse dust (McK95).

E.5.7 Chromium

From a theoretical viewpoint, chromium would be expected to remain primarily in the melt. However, Stubbles suggests that chromium recovery in the melt during EAF steelmaking is only 30 to 50% (STU84a). Stubbles' observation is not consistent with the calculations in Table E-1, which show Cr remaining primarily in the melt.

Xiao and Holappa have studied the behavior of chromium oxides in various slags at temperatures between 1,773 K and 1,873 K (XIA93). They reported that chromium in the slag was mainly (*i.e.*, 88% to 100%) Cr^{+2} when the mol% CrO_x in the slag was 10% or less and the N_{CaO}:N_{SiO2} ratio was unity. The calculations in Table E-1 assumed Cr^{+3} to be the predominant species. Using free energy data from XIA93 for the reaction:

$$Cr_{(s)} + \frac{1}{2}O_2 = CrO_{(l)}$$

(*i.e.*, $\Delta F° = -79{,}880 + 15.25T$ cal) and other relevant data from Table E-1, the partition ratio involving CrO rather than Cr_2O_3 is calculated to be 0.42. This suggests that a significant portion of the Cr will partition to the slag if Cr^{+2} is the principal cation in the slag.

E.5.8 Cobalt

Free energy calculations indicate that Co should remain primarily in the melt. Nakamura and Fujuki found this to be the case in 500-kg air induction melts of carbon steel and stainless steel where Co-60 was detected only in the ingots (NAK93). During the melting of six heats of contaminated carbon steel scrap at INEL some (unquantifiable) Co-60 activity was detected in the dust collection system and some in the slag (LAR85a). In subsequent tracer tests with three heats of Type 304L stainless steel, between 96 and 97% of the Co-60 was recovered in the ingots (LAR85b). Sappok *et al.* noted that, during the induction melting of steel, Co-60 was mostly found in the melt although unquantifiable amounts were detected in the slag and in the dust collection system (SAP90). In an earlier paper, Sappok cited the Co-60 distribution from nine melts totaling 24 metric tons as 97% in the steel, 1.5% in the slag and 1.5% in the cyclone and baghouse (PFL85). Schuster and Haas measured the Co-60 distribution in laboratory melts of St37-2 steel and reported 108% in the ingot, 0.2% in the slag and 0.2% in the aerosol filter (SCH90).

According to Harvey (HAR90), " ...cobalt 60 will almost certainly be retained entirely in the steel in uniform dilution in both electric arc and induction furnaces." In support of this conclusion, Harvey described two steel melts in a 5-ton electric arc furnace. In one test, highly reducing conditions were employed (high carbon and ferrosilicon) while, in the other, the conditions were oxidizing (oxygen blow). In neither case was any measurable Co activity found in the slag. The amount of Co-60 found in the melt was in good agreement with the amount predicted from the furnace charge. No Co-60 was found in the furnace dust although some was expected based on transfer of slag and oxidized steel particles to the gas cleaning system. Harvey concluded that the low level of radioactivity in the furnace charge (*ca.* 0.23 Bq/g) coupled with dilution from dust already trapped in the filters resulted in quantities of Co-60 in the off-gas below the limits of detection.

Menon commented on the air induction melting of 33.6 tons of carbon steel. No Co-60 was detected in the slag, but a small quantity (1,300 Bq/kg) was detected in the baghouse dust. The amount remaining in the ingots was not quoted. In two heats of stainless steel weighing a total of 5,000 kg, 26 x 10^6 Bq of Co-58/Co-60 were measured in the ingots, 4 x 10^4 Bq in the slag and 7.8 x 10^4 Bq in the baghouse dust (MEN90).

E.5.9 Europium

Based on its chemical similarity to other rare-earth elements such as samarium, cerium and lanthanum, europium is expected to partition to the slag. During induction melting of steel scrap from nuclear installations, Sappok reported that all the Eu-154 was in the slag (SAP90). Larsen found some Eu in the slag and some in the baghouse dust during induction melting of scrap from the SPERT III reactor. The Eu content was below the limits of detection in the feed material, so presumably some unquantified concentrating effects occurred in the slag and the off-gas dust (LAR85a). Eu-152 concentrations in the baghouse dust were very low—on the order of 0.8 pCi/g. Harvey described production of an experimental 3,500 kg melt of steel in an arc furnace to study europium partitioning (HAR90). During the melting operation, oxygen was blown into the melt to remove 0.2% C (typical of normal steelmaking practice). The radioactivity of the metal was too low to be measured and no europium was found in the dust from the fume extraction system. Europium activity was detected only in the slag. Even though there was some concern expressed that, because of the similar densities of steel and europium oxide (7.9 g/cm^3 and 7.4 g/cm^3, respectively), the europium oxide would not readily float to the metal/slag interface, the experimental results suggest this was not an issue. With regard to the fact that no europium was found in the fume collection system, Harvey observed (HAR90):

> It is inevitable, however, because of the nature of the process, that some slag is ejected into the atmosphere of the arc furnace and is then entrained in the off-gas and is collected in the gas cleaning filters. Hence any radioactive component present in the slag will be present to some extent in the off-gas. The fact that it is not detected on this occasion reflects the small amount of radioactivity used, and the mixing and dilution of dust which occurs in the gas cleaning plant.

E.5.10 Hydrogen

Hydrogen is an undesirable impurity in steel causing embrittlement. Thus steelmaking practice seeks to keep the contaminant at very low levels. As noted in Section E.5.3, removal of charge carbon by blowing oxygen through the melt reduces the hydrogen as well. Stubbles described tests on the rate of hydrogen removal as a function of time and carbon reduction rate (STU84b). For steel with an initial hydrogen content of 9 ppm, the hydrogen level was reduced to 1 ppm after 15 minutes when the rate of carbon removal was 1% per hour and to 5 ppm over the same interval when the carbon removal rate was 0.1% per hour.

Stubbles' work is consistent with results reported by Deo and Boom (DEO93) who showed that the rate of hydrogen removal was directly related to the rate of carbon removal. They also described the work of Kreutzner (KRE72) who investigated the solubility of hydrogen in steel at 1,873 K and 1,973 K. From a graphical presentation of Kreutzner's work, one can estimate that the solubility of hydrogen in steel at 1,873 K can be expressed as

$$[H] = 27\, P_{H_2}^{1/2}$$

where [H] is the hydrogen solubility in ppm and P_{H_2} is the hydrogen partial pressure in atmospheres. Thus, when P_{H_2} is 0.01 atm, the eqillibrium hydrogen concentration is 2.7 ppm.

Since the most likely source of hydrogen is from water in the charge components or the furnace atmosphere, the following reaction should also be considered (PHI51):

$$H_2O_{(g)} = 2\underline{H} + \underline{O}$$

At 1,873 K, the equilibrium hydrogen concentration is

$$\%\underline{H} = 1.35\cdot 10^{-3} \left(\frac{P_{H_2O}}{a_O} \right)^{1/2}$$

where a_O is the activity of oxygen in the melt. One can see from this equation that the %$\underline{H}$ increases as a_O decreases. When P_{H_2O} is 0.003 atm, concentrations of $\underline{H}$ are as follows, for various assumed dissolved oxygen concentrations:

Concentration (%)	
$\underline{O}$	$\underline{H}$
0.1	2.5E-4
0.01	8E-4
0.001	2.5E-3

If the oxygen content of the bath is low, the steel can absorb more hydrogen from water vapor than from pure hydrogen at 1 atm. Hydrogen or water vapor in materials added to the bath after carbon removal or in the furnace ladle will tend to be retained in the product steel (PHI51).

E.5.11 Iridium

Iridium would be expected to remain in the melt during steelmaking. Iridium and iron are completely miscible in the liquid phase (ASM93). INEL conducted one induction melting test at the Waste Experimental Reduction Facility (WERF) where Ir-192 was added to Type 304L stainless steel to produce about 500 lb of product. About 60% of the charged iridium was recovered in the ingot but only small quantities were detected in the slag. Although the material balance was poor, there is no basis to conclude that iridium does not primarily remain in the melt (LAR85b).

E.5.12 Iron

Iron oxide is a major slag component. According to a 1991 survey by the National Slag Association, the average FeO content of steel slags is 25% (NSA94). If one assumes that the ratio of slag mass to steel mass is 0.1, then about 2% of the iron in the charge would be distributed to the slag. Schuster *et al.* reported some laboratory tests where Fe-55 was added to small melts of steel conducted under an Ar + 10% H_2 atmosphere and reducing conditions (SCH90, SCH88). No Fe-55 was found in the slag or the aerosol filter. However, these results have little relevance to expected partitioning under actual steelmaking conditions.

E.5.13 Lead

As shown in Table E-1, lead should remain with the melt rather than with the slag. At 1,873 K, lead has limited solubility in molten iron—about 0.064 to 0.084 wt% (ASM93). Although the boiling point of lead (2,010 K) is above normal steelmaking temperatures, lead has a significant vapor pressure (*ca.* 0.4 atm) at 1,873 K. In addition, any PbO which forms during initial heating of the furnace charge could volatilize before the steel begins to melt since PbO is a stable gas at steelmaking temperatures (GLA57, KEL66). Consequently, much of the lead should be transferred from the melt either as lead vapor or as gaseous lead oxide and be collected in the off-gas system. Stubbles reports that, when leaded scrap is added to liquid steel, the lead boils off like zinc and is collected with the fume (STU84a). If lead in the form of batteries or babbitts is added to the furnace charge, the lead will quickly melt and sink to the bottom of the furnace where it may penetrate the refractory lining.

E.5.14 Manganese

Manganese is a common element in steelmaking. As discussed above, a typical carbon steel contains 0.6 to 0.9% Mn. Calculations in Section E.2 show that manganese should be more concentrated in the slag than in the metal. For EAF melting, Stubbles states that about 25% of the Mn is recovered in the steel. This establishes the partition ratio based on the mass of Mn in slag to the mass of Mn in steel at 3:1.

Meraikib complied information on manganese distribution between slag and molten iron based on a large number of heats in a 70-ton electric arc furnace (MER93). He showed that the ratio of the concentration of manganese in the slag to manganese in the metal, η_{Mn}, is given by the following equation:

$$\eta_{Mn} = \frac{(Mn)}{[Mn]}$$

$$= a_{[O]} f_{[Mn]} \exp\left(\frac{27530}{T} - 0.0629\,B - 7.3952\right)$$

where:

(Mn) = concentration of Mn in slag (wt%)
[Mn] = concentration of Mn in melt (wt%)
$a_{[O]}$ = activity of oxygen in melt
$f_{[Mn]}$ = activity coefficient for [Mn]

All other terms have been defined previously.

For the range of manganese concentrations (0.06 to 1.0 wt%) and the range of temperatures (1,823 K to 1,943 K) studied, $f_{[Mn]}$ is essentially unity (*i.e.*, 0.9503). If one assumes that $B = 2$ and $a_{[O]} = 0.004$, then the variation of η_{Mn} with temperature can be calculated as follows:

1,843 K .. $\eta_{Mn} = 6.3$
1,943 K .. $\eta_{Mn} = 2.9$

indicating that the concentration ratio of manganese between slag and metal can vary by a more than factor of two for a 100 K change in melt temperature. Based on the work of Meraikib, the partitioning of Mn between slag and metal (assuming a slag:metal ratio of 1:10) is an order of magnitude lower than observed by Stubbles and about two orders of magnitude lower than estimated from thermodynamic principles in Section E.2 This suggests that the oxygen activity in the steel in equilibrium with the slags used in Meraikib's work is lower than implied in the free energy calculations in Section E.2

Nakamura and Fujuki conducted four 500-kg air-induction melting tests (two with ASTM-A335 steel and two with SUS 304 stainless steel) to which 24 MBq of Mn-54 were added (NAK93). In two tests with SUS 304 and one test with ASTM-A335, about 90% of the radioactivity was contained in the ingot, while in the other ASTM-A335 ingot only 50% of the Mn-54 was recovered. For the one ASTM-A335 ingot where the slag concentration was also reported, the distribution based on input radioactivity was:

- ingot 91%
- slag 8%
- unaccounted .. 2%

Sappok *et al.* described experience in melting about 2,000 tons of contaminated steel in a 20-ton induction furnace (SAP90). The melting process generated only a small amount of slag (*i.e.*, about 1.2%). During a 200-ton melting campaign, no Mn-54 was found in the melt. Up to 21.9% of the total slag activity was attributed to Mn-54 and up to 2.1% of the total activity in the dust collection system was from this nuclide.

Harvey notes that Mn tends to be more concentrated in the slag when melting under oxidizing conditions although the reverse result can be obtained when the furnace conditions are reducing (HAR90). Manganese is relatively volatile having a vapor pressure of 0.08 atm at 1,900 K.

In two stainless steel heats melted at Studsvik, the combined manganese distribution was (MEN90):

- Ingot .... 4.4×10^4 Bq
- Slag .... 3.6×10^3 Bq
- Baghouse dust .... 3.6×10^2 Bq

E.5.15 Molybdenum

As described previously in Section E.2, Mo should remain primarily in the melt. Stubbles supports this view indicating that 100% of Mo is recovered in the steel during electric arc furnace melting (STU84a). Studies by Chen on the reduction kinetics of MoO_3 in slag also buttress this conclusion (CHE93). In 1-kg scale laboratory tests, Chen found that the reduction of MoO_3 in slag over an iron-carbon melt was completed in about five minutes.

E.5.16 Nickel

Nickel is chemically similar to Co and should remain in the melt during steelmaking. Stubbles states that nickel recovery during arc melting is 100% (STU84a). According to Harvey, it is common practice to add nickel oxide to a steel melt and quantitatively recover the nickel. He further notes: "Nickel cannot be volatilized from molten steel, and there do not appear to be any slags which will absorb nickel selectively." (HAR90). Schuster described the distribution of Ni-63 in laboratory melts of 3 to 5 kg under inert gas (SCH90). About 82% of the nickel was recovered in the ingot, 0.04% in the slag and 0.06% in the aerosol filter, with the remainder unaccounted for.

E.5.17 Niobium

On the basis of the thermodynamic calculations in Section E.2, niobium should partition primarily to the slag. According to Stubbles, the recovery of niobium from scrap in the ingot is zero during EAF melting, which is consistent with the theoretical calculations (STU84a). Harvey (HAR90) notes that Nb can be retained in the steel under reducing conditions but under oxidizing conditions will clearly be transferred to the slag according to the reaction:

$$2\underline{Nb} + 6\underline{O} + Fe = FeO{\cdot}Nb_2O_5$$

The equilibrium constant for this reaction is :

$$K_1 = \frac{a_{FeO\cdot Nb_2O_5}}{a_{\underline{Fe}}\; a_{\underline{Nb}}^{2}\; a_{\underline{O}}^{6}}$$

indicating that the equilibrium is very sensitive to the activity of the oxygen in the steel. At 1,873 K, $K_1 = 2.4 \times 10^{10}$.

Wenhua *et al*. studied the kinetics of Nb_2O_5 reduction in slag by silicon dissolved in iron (WEN90) according to the reaction:

$$5\underline{Si} + 2(Nb_2O_5) = 4\underline{Nb} + 5(SiO_2)$$

The reaction was assumed to be divided into five steps:

1. Nb_2O_5 diffuses through slag towards reaction interface
2. Si diffuses through molten iron towards reaction interface
3. Reaction occurs at interface
4. Reaction product Nb diffuses from interface into molten iron
5. Reaction product SiO_2 diffuses from interface into slag

Using a slag with a CaO:SiO_2 (basicity) ratio of about 2:1 and a ferrosilicon reductant (*ca* 0.42% Si), Nb was rapidly transferred from the slag to the melt reaching a value of 1.5% after 10 minutes. Wenhua found that the rate controlling step was the diffusion of Nb in liquid iron.

E.5.18 Phosphorus

Phosphorus is an undesirable impurity in steel which is typically removed by oxidation. The transfer of phosphorus from the metal to the slag can be represented by the following simplified reaction (STU84b):

$$2\underline{P} + 5\underline{O} = (P_2O_5)$$

The amount removed from the melt will depend on the P content of the scrap charge and the desired P content of the melt. Phosphorus removal is facilitated during EAF melting by increasing the basicity and oxidation level of the slag. By injecting 35 kg of powered lime per ton into the melt together with oxygen, the phosphorus content can be reduced to about 10% of its initial value.

E.5.19 Potassium and Sodium

Since K_2O is less stable than FeO, potassium should be removed from the melt because of its low boiling point. However, various potassium compounds such as silicates and phosphates are present in slags (HAR90). The same considerations apply to sodium. Na_2O has also been collected in electric arc furnace baghouse dust (BRO72). Given the fact that Na_2O in the slag can be reduced by carbon in the melt (MUR84), that observation is not surprising. The appropriate chemical equation is:

$$Na_2O_{(l)} + \underline{C} = 2Na_{(g)} + CO_{(g)}$$

ΔF° for this reaction at 1,873 K is -48,000 cal/mole. Removal of Na_2O from the slag would be enhanced by higher carbon levels in the melt. Presumably any Na from this reaction would be vaporized and subsequently condensed in the baghouse as Na_2O.

E.5.20 Plutonium

Thermodynamic predictions suggest that plutonium will partition strongly to the slag. Harvey assumed, based on the chemical similarity of plutonium with thorium and uranium, that the plutonium will form a stable oxide and be absorbed in the slag (HAR90). However, he notes that because of its high specific gravity (11.5), transfer of PuO_2 to the slag could be slow and some could possibility fall to the base of the furnace and not reach the slag.

Gerding *et al.* conducted small-scale (*i.e.*, 10 g and 200 g) tests with plutonium oxide and mild steel in an electric resistance furnace (GER77). The melts were held in contact with various slags for 1 to 2 hours at 1,773 K under He at about 0.5 atm. Slag:steel weight ratios ranged from 0.05 to 0.20. The studies showed that the Pu partitioned to the slag and the partition coefficients (concentration in slag ÷ concentration in metal) were 2×10^6 to 8×10^6. Decontamination efficiency was about the same at 400 and 14,000 ppm Pu, and differences in composition among the various silicate slags were not significant to the partitioning.

E.5.21 Radium

Radium forms a stable oxide in the presence of FeO and thus would be expected to be found mainly in the slag. Starkey described results from the arc furnace melting of eight heats of steel contaminated with Ra (STA61). The average concentration of the Ra in the steel was $<9 \times 10^{-13}$ g Ra/g steel and in the slag was 1.47×10^{-9} g Ra/g slag. Slag/metal mass ratios were not reported, but assuming the mass slag/mass metal is 0.1, then the partitioning ratio (mass Ra in slag/mass of Ra in metal) is >160.

E.5.22 Silver

As noted in Section E.2, silver will not react with FeO because Ag_2O is unstable at steelmaking temperatures. Silver has no solubility in liquid iron and thus the two metals will coexist as immiscible liquids (ASM93). Since silver has a significant vapor pressure (*ca.* 10^{-2} atm at 1,816 K), some volatilization might be expected. Sappok reported that induction melting of steel contaminated with silver resulted in the silver being primarily distributed to the metal, but some was detected both in the slag and in the off-gas dust (SAP90). However, the distribution was not quantified. Harvey concluded, based on the instability of Ag_2O and the expected similarity to the behavior of copper in steel, that silver "would be expected to remain in the melt under all normal steelmaking conditions." (HAR90).

Ag-110m activity was measured for two heats of stainless steel by Studsvik (MEN90). The Ag-110m activity was distributed as follows:

- Ingot .... 2.9×10^5 Bq
- Slag .... 1.3×10^3 Bq
- Baghouse dust .... 9.3×10^4 Bq

E.5.23 Strontium

Strontium is predicted to partition to the slag. Nakamura and Fujuki studied the partitioning of Sr-85 during the air induction melting of ASTM-A335 steel in a 500-kg furnace with a slag basicity of 1 (NAK93). All of the Sr-85 was found in the slag (recovery was 75%). Larsen *et al.* described the melting of three heats of Type 304L stainless weighing 500 to 700 lb each in an air induction furnace (LAR85b). The amount of Sr remaining in the ingots was 1% in two cases and 0 in the third. Sr-85 was found in the slag and the baghouse dust but no mass balance was provided. Slagging practice was not documented other than to state that a small amount of a "slag coagulant" was added to aid in slag removal. Schuster and Haas melted St37-2 steel in a 5-kg laboratory furnace using a carborundum crucible. Lime, silica and alumina were added as slag formers. The melt was allowed to solidify *in situ*. About 80% of the Sr-85 was found on the ingot surface, 6.3% in the slag, 0.5% in the ingot and 0.02% in the aerosol filter. The material on the ingot surface would most likely have been found in the slag under more realistic production conditions.

Strontium can also react with sulfur and the resultant SrS should partition to the slag (BRO85).

E.5.24 Sulfur

Sulfur is a generally undesirable element except in certain steels where higher sulfur levels are desired for free machining applications. As indicated at the beginning of this section, the maximum sulfur content of a typical low carbon steel is 0.05%. Sulfur is difficult to remove from the melt. One mechanism for sulfur removal is reaction with lime in the slag to form calcium sulfide according to the reaction:

$$CaO + \underline{S} = CaS + \underline{O}$$

This reaction is facilitated by constant removal of high basicity slag and agitation. According to Stubbles, the concentration ratio $\frac{(S)}{[S]}$ rarely exceeds 8 in EAF melting of steel (STU84b). Although sulfur has a very low boiling point (see Table E-3), the compounds it forms within the slag (*e.g.*, CaS) are very stable at steelmaking temperatures.

Engh described the partitioning of sulfur between slag and metal as a function of slag acidity and FeO content of the slag (ENG92). Assuming that the slag contained 25% FeO and 20% acid components (SiO_2, P_2O_5, B_2O_3 and TiO_2), the ratio $\frac{(S)}{[S]}$ would range between about 16 and 26.

E.5.25 Thorium

Based on the stability of ThO_2, thorium should partition to the melt. Harvey notes that the stability of ThO_2 has been exploited by using the material in steel melting crucibles (HAR90). However, because of their high specific gravity (9.86), ThO_2 particles may settle in the melt and not reach the slag.

E.5.26 Uranium

Free energy calculations suggest that uranium should partition to the slag. Heshmatpour and Copeland conducted a number of small-scale partitioning experiments where 500 to 1,000 ppm of UO_2 was added to 50 to 500 grams of mild steel and melted in either an induction furnace or a resistance furnace. Slag and crucible composition were varied as well (HES81). With the use of highly fluid basic slags and induction melting, partition ratios (mass in slag:mass in metal) from 1.2:1 to >371:1 were obtained.

Larsen reported that; although U was not detected in the feed stock, it was sometimes found in the slag and in the baghouse dust (LAR85a). Schuster and Haas determined in small laboratory melts that when slag formers were added, the U content was reduced from 330 μg U/g Fe to 5 μg U/g Fe (SCH90). Harvey commented that British Steel had occasionally used uranium as a trace element in steelmaking (HAR90). Based on their experience, the uranium was absorbed in the slag in spite of the fact that UO_2, which has a density (10.9 g/cm^3) significantly higher than that of iron, could conceivably settle in the melt.

Abe *et al.* studied uranium decontamination of mild steel using small (100 g) melts in a laboratory furnace (ABE85). Melting was done in an argon atmosphere at a pressure of 200 torrs in alumina crucibles with 10 wt% flux added to the charge. The uranium decontamination factor was found to be a function of the initial contamination level, varying from about 200 to about 5,000 as the uranium concentration increased from 10 to 1,000 ppm. Optimum decontamination occurred when the slag basicity was 1.5 with a $CaO\text{-}Al_2O_3\text{-}SiO_2$ slag. Decontamination was further enhanced by additions of CaF_2 or NiO to the slag.

E.5.27 Zinc

Zinc is not expected to react with the slag consituents and, because of its low boiling point, some fraction should evaporate from the melt. In fact, dust from steelmaking operations is an important secondary source of Zn. In 1990, about 100,000 tons of zinc were recovered from baghouse dust in Europe (PER92). Hino *et al.* studied the evaporation of zinc from liquid iron at 1,873 K and found that the evaporation rate was first order with respect to the zinc content of the melt (HIN94). The mass transfer coefficient in the liquid phase was estimated to be 0.032 cm/s.

Nakamura and Fujuki observed that, when induction melting both ASTM-A335 and SUS 304 steels, about 60% to 80% of added Zn-65 remained in the ingot (NAK93). In one test with ASTM-A335 steel, 90.7% of the added Zn was recovered. Of the total amount recovered, about 14% was found in the off-gas and 1% in the slag with the balance remaining in the ingot. Sappok reported that, in some instances, zinc was found only in the off-gas collection system and, in another melting campaign, some zinc was found in the ingot and the slag as well as in the off-gas system. The causes of these differences are not apparent (SAP90).

On the other hand, Stubbles states that Zn is volatilized during EAF melting (STU84a). Harvey supports the view of Stubbles noting that zinc is volatilized during melting and collected as zinc oxide in the baghouse filters (HAR90). "The volatilization is very efficient, and the residual content of zinc in the steel is likely to be below 0.001%, whereas the zinc oxide content of the dust is often more than 10%."

Perrot *et al.* note that in spite of its low boiling point and expected ease of evaporation, zinc removal from liquid steel is far from complete (PER92). Industrial experience indicates that the zinc content is often above 0.1 wt.% in liquid cast iron at 1,573–1,673 K but is somewhat lower in liquid steel at 1,773–1,873 K. At 1,773 K, assuming that the zinc vapor pressure over the melt is 0.01 atmosphere, the calculated solubility of zinc in iron is about 72 ppm. The solubility of Zn in liquid iron is decreased by other solute elements with ion interaction coefficients greater than zero (*e.g.*, Al and Si) and decreased by solutes with coefficients less than zero (*e.g.*, Mn and Ni).

Richards and Thorne studied the activity of ZnO in slags with various $CaO:SiO_2$ ratios, over the temperature range 1,373 to 1,523 K, based on the assumption that the following slag/metal reaction controlled the equilibrium:

$$(ZnO) + Fe_{(s)} = (FeO) + Zn_{(g)}$$

The parentheses indicate slag components, as usual. Further assuming that the gas phase contained 3 vol% Zn, they calculated that, at 1,473 K, the amount of Zn in the slag could be represented by the expression:

$$(\text{wt\% Zn}) = \frac{0.022\ (\text{wt\%FeO})\ (\gamma_{FeO})}{(\gamma_{ZnO})}$$

where all components of the equation involve the slag phase. For a fixed FeO concentration, the amount of Zn in the slag decreased with increasing temperature and increasing ratios of $CaO:SiO_2$. For example, at 1,473 K, when the $CaO:SiO_2$ ratio was 0.3:1, the slag contained 1.2 wt% Zn and, when the $CaO:SiO_2$ ratio was 1.2:1, the Zn content of the slag had dropped to 0.8 wt%. If one extrapolates these results to 1,873 K, the amount of Zn in the slag would be only about 0.009%.

Menon found that, during the melting of two stainless steel heats, the Zn-65 was about equally distributed between the melt and the baghouse dust (MEN90).

From the available information it appears that, when the scrap metal charge has a reasonably high zinc content, significant amounts of zinc will be volatilized but, when the zinc levels in the charge are low, vaporization will be more difficult. Virtually no zinc should remain in the slag.

E.5.28 Zirconium

Based on free energy considerations, Zr would be expected to partition to the slag. Stubbles' information for EAF steel melting supports this hypothesis (STU84a).

E.6 INFERRED PARTITIONING

No theoretical or experimental evidence exists for the partitioning of several elements that may be contaminants in steel. This section proposes the distribution of these nuclides based on chemical and/or physical behavior.

E.6.1 Curium

Curium should behave like other elements in the actinide series such as americium and partition to the slag.

E.6.2 Promethium

Promethium should behave like other rare-earth elements such as europium and samarium and partition to the slag.

E.7 SUMMARY

In summarizing the distribution of the various potential contaminants that might be introduced into the steel melting process, one must define certain process parameters including:

- ratio of mass of steel produced to total mass of scrap charged—imported scrap + home scrap (R_1)
- ratio of mass of slag to mass of steel produced (R_2)
- ratio of mass of baghouse dust to mass of steel produced (R_3)
- fraction of baghouse dust from slag (%Sl)
- fraction of baghouse dust from steel (%St)

The following values were adopted for each of these process parameters:

- R_1 ...... 0.9 [12]
- R_2 ...... 0.13 [13]
- R_3 ... 30 lbs/ton of steel melted (33 to 36 pounds per ton of carbon steel produced in EAF—ADL93)[14]
- %Sl ... 33.3 [15]
- %St ... 66.7

The R_1 value is based on the following assumptions:

- 5% of metal in each heat becomes home scrap, which is returned to the furnace in a later heat
- 1.5% of metal is lost to baghouse dust
- 2% of metal is lost to slag
- 1.5% is unaccounted for

Based on these process parameters and the information presented previously, the assumed distribution of the various elements in summarized in Table E-6. Since the amount of baghouse dust contributed by the melt is 20 lb/ton, if a potential radioactive contaminant tended to concentrate in the melt, the dust would contain 1% of the activity in the melt. Similarly, since the amount of baghouse dust contributed by the slag is 10 lb per ton of metal, and since the mass of the slag is $\frac{1}{10}$ the mass of the melt, if such a contaminant tends to concentrate in the slag, 5% of the slag activity would be transported to the baghouse. For simplicity, the baghouse efficiency is assumed to be 100% in evaluating partition ratios.

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[12] Bayou Steel states that they typically produce 0.882 tons of steel billets per ton of scrap charged (private communication with Al Pulliam, June 25, 1996). When averaged over the total U.S. production, the process efficiency is much higher. According to the U.S. Geological Survey for the year 1994, the amount of recirculating home scrap was 132,300 tons, while 39.5 million tons of EAF steel were produced. Thus, the annual average ratio of home scrap to steel produced was 0.3% (private communication with M. Fenton, June 25, 1995).

[13] According to R. West of International Mill Services, a major slag marketer, between 0.12 and 0.14 tons of slag are generated per ton of steel produced (private communication - June 25, 1996). Since this appears to be a more realistic figure than the 10% cited in STU84a, the average of 0.13 was adopted for the present analysis.

[14] Additional information on baghouse dust is included in Appendix E-2.

[15] Based on the baghouse dust composition reported by SAIC (McK95) adjusted for the ZnO content and assuming that all the Fe_2O_3 and half the MnO and SiO_2 are from the melt, the %Sl is 33%.

Where varying results are presented by different investigators, emphasis was placed on results which represented EAF melting of carbon steel with basic slags.

Considerable care must be used in interpreting the experimental results cited in Section E.5 and applying them to predicting contaminant distributions during the EAF melting of carbon steel. Some concerns are summarized below.

- In many cases, the results are based on induction melting which is a more quiescent process than arc melting. Agitation of the slag and melt should tend to drive reactions toward equilibrium.

- Often, the slag chemistry was either not cited or no attempt was made to optimize the slag-metal reactions as required in commercial melting practice.

Table E-6. Proposed Distribution of Potential Contaminants During Carbon Steelmaking

Element	Distribution (%)				Comments
	Melt	Slag	Baghouse	Atmosphere	
Ac		95	5		
Ag	99/75		1/25		
Am		95	5		
Ba		95	5		
Bi			100		Assumed same as Pb
C	100/27			0/73	Depends on melting practice
Ca		95	5		
Cd			100		
Ce		95	5		
Cl		50	50		Some Cl in baghouse dust (McK95)
Cm		95	5		
Co	99		1		
Cr	99/40	0/57	1/3		Longest-lived isotope: $t_{1/2} = 27.7$ d
Cs		0/5	100/95		
Cu	99		1		Longest-lived isotope: $t_{1/2} = 2.58$ d
Eu		95	5		
Fe	97	2	1		

Table E-6. Proposed Distribution of Potential Contaminants During Carbon Steelmaking (Continued)

Element	Distribution (%)				Comments
	Melt	Slag	Baghouse	Atmosphere	
H	10			90	Needs further analysis
I				100	
Ir	99		1		
K		50	50		Needs further analysis
Mn	24/65	72/32	4/3		
Mo	99		1		
Na		50	50		Needs further analysis
Nb		95	5		
Ni	99		1		
Np		95	5		
P	9	87	4		Longest-lived isotope: $t_{1/2} = 25.3$ d
Pa		95	5		
Pb			100		
Pm		95	5		
Po			100		
Pu		95	5		
Ra		95	5		
Re	99		1		
Rn				100	
Ru	99		1		
S	19	77	4		Slag % is max. expected. Melt % may be higher. (Maximum $t_{1/2} = 87.2$ d.)
Sb	99/80		1/20		Conflicting reports on Sb distribution
Se	19	77	4		Assumed to behave like S
Sm		95	5		
Sr		95	5		
Tc	99		1		
Th		95	5		
U		95	5		
Y		95	5		
Zn	20/0		80/100		Zn difficult to remove from melt at low concentrations
Zr		95	5		

Table E-6 (continued)

- In some cases, results are quoted for stainless steels rather than carbon steels. The thermodynamic activity of solutes in the highly alloyed steel melt should be different from that in plain carbon steels and the slag chemistry will be significantly altered.

- Perspective on kinetically driven processes may be altered by the scale of the melting operation.

- Melt temperatures and holding times in the molten state may be quite different in cited experiments as compared to commercial practice. This can significantly impact conclusions, especially with regard to volatile elements. The mass concentrations of potential contaminants in free-released steel scrap would be quite low. Consequently, some of the partition predictions made here may be overridden by other factors. For example, if evaporation kinetics of volatile elements control the release, small quantities of zinc may remain in the steel. For strong oxide formers which should partition to the slag, transfer may be impeded due to the high density of many of the actinide and rare-earth oxides. The experimental evidence of this possibility is mixed. For example, Eu_xO_y seems to be removed from the melt during normal electric arc furnace melting, but CeO_2 may not be completely removed. One investigator reported that the uranium decontamination factor in mild steel increased with increasing contaminant levels (ABE85).

In addition, the expected partitioning may be altered significantly if the melting practice is changed. Examples presented in this report include the removal of Nb from the slag to the melt and movement of W in the opposite direction.

The information in Table E-6 does not explicitly consider home scrap or contaminated furnace refractories. Home scrap (*i.e.*, the scrap from the melting process that is recirculated into future furnace charges) should have the same contaminant distribution as the melt from which it was produced. The contamination of furnace refractories was not studied in this report. However, it should be noted that residuals remaining in the furnace from a melt are frequently recovered in the next one to two melts. For example, when melting a low alloy steel containing, say, 1% Cr, the following heat or two will contain more Cr than would be expected if the only source were the furnace charge for the ensuing heats.[16]

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[16] Private communication with J. R. Stubbles, Charter Steel Company - July 1, 1996.

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APPENDIX E-1

EXTENDED ABSTRACTS OF SELECTED REFERENCES

CHE93 Chen W., *et al.*, "Reduction Kinetics of Molybdenum in Slag," in *Steel Research,* vol. 63, No. 10, pp. 495-500, 1993.

Reduction of molybdenum oxide in slag over an iron-carbon melt is completed in 5 min in 1-kg lab melts.

The reaction may be:

$$(MoO_3) + 3[C] = [Mo] + 3CO_{gas}$$
$$\Delta F^\circ = 82.35 - 0.2370T \text{ [kJ]}$$

or a two-step process

$$(MoO_3) + 3Fe = [Mo] + 3FeO$$
$$\Delta F^\circ = -213.6 + 0.0386T \text{ [kJ]}$$

and

$$(FeO) + [C] = Fe + CO_{gas}$$
$$\Delta F^\circ = 98.65 - 0.0919T \text{ [kJ]}$$

At 1,440 to 1,500°C the reaction rate is controlled by Mo diffusion in slag and, from 1,500 to 1,590°C, the reaction rate is controlled by Mo diffusion in the melt.

GOM85 Gomer, C. R., and J. T. Lambley, "Melting of Contaminated Steel Scrap Arising in the Dismantling of Nuclear Power Plants," British Steel Corporation, for Commission of the European Communities, Final Report Contract No. DED-002-UK, 1985

This paper discusses the same tests but in somewhat greater detail than Pflugard *et al.* (PFL85). The electric arc furnace slag is about 5% to 10% of the metal cast weight and involves chiefly additions of carbon, lime and ferrosilicon plus eroded refractories and general oxidation products. Melts were about 2.5 tons each. In the arc furnace melt with a CsCl addition, Cs was added with melt charge and since CsCl is volatile below steelmaking temperature, the CsCl volatilized before any could be incorporated into non-reactive basic slag. In an induction furnace test, *CsOH was added into liquid steel pool with complete cover of relatively cool, quiescent acid slag*. In an arc furnace test with CsOH, Cs was added to molten pool but slag conditions are not described nor is the hold time after addition stated. However, Gomer stated that, although the slag was made as acidic as the furnace liner could withstand, it still did not contain enough silica to fix the cesium as cesium silicate. The limited Cs recovery of only 50% was attributed to Cs condensation on cooler duct walls upstream of sampling point. In an arc furnace test with cesium sulfate, Cs was added as in the previous arc furnace test with CsOH. The higher Cs recovery in the slag is attributed to incorporation of cesium sulfate into the slag.

LAR85a Larsen, M. M., *et al.*, "Sizing and Melting Development Activities Using Contaminated Metal at the Waste Experimental Reduction Facility," EGG-2411, EG&G Idaho, Inc. February 1985.

This report describes melting of contaminated carbon steel from the SPERT III reactor in a 1,500 lb coreless induction furnace at the Waste Experimental Reduction Facility (WERF). Six heats were thoroughly sampled. All showed only Co-60 in feed stock. However, due to concentrating effects, Eu, Cs and occasionally U were found in the slag, while the baghouse dust contained Co, Cs, Eu and U, and spark arrestor dust contained Co and Eu. This occurred even though, except for Co-60, all these nuclides were not seen in the feed at the limits of detection. Molten metal samples either contained Co-60 or emitted no detectable radiation.

Detectable quantities of Co-60 were seen in slag and baghouse and spark arrestor dust. Of 35,900 Ci of Co-60 charged into six melts, 1,361 Ci were recovered in the baghouse and spark arrestor dust (3.8%).

LAR85b Larsen, M. M., *et al.*, "Spiked Melt Tests at the Waste Experimental Reduction Facility," PG-WM-85-005, Idaho National Engineering Laboratory, EG&G Idaho Inc., February 1985.

Tracer tests were conducted at WERF in a 1500 lb induction furnace using Type 304L stainless steel. Three heats, weighing 474 to 689 pounds each, were made. All were doped with Co-60, Cs-137 and Sr-85, while Ir-192 was added to only one. Melt temperatures were not specified; slag chemistry was not specified but apparently no slag formers were added.[17] A small amount of slag "coagulant" was added to aid in slag removal. Tracers were added to the initial furnace charge.

Based on *melt* samples, the following percentages remained in the ingots:

Test 1: Co-60 87%, Cs-137 1.3%, Sr-95 1.7%
Test 2: Co-60 73%, Cs-137 1.8%, Sr-85 2.3%
Test 3: Co-60 77%, Cs-137 1.8%, Sr-85 2.3%, Ir-192 57%

Subsequent analysis of the *ingots* suggested that these analyses were biased low because of the large sample sizes taken from the melts which caused self-shielding. Averaged results from ingot tests as follows are believed to be more reliable (avg. % isotope remaining in ingot):

- Test 1: Co-60 96%, Cs-137 10%, Sr-85 1%
- Test 2: Co-60 96%, Cs-137 8%, Sr-85 0%
- Test 3: Co-60 97%, Cs-137 5%, Sr-85 1%, Ir-192 60%

The fraction of the charge recovered in the ingot was 93% in Test 1, 98.4% in Test 2 and 95.4% in Test 3.

Some problems were encountered with entrained metal in the slag samples. Poor results were obtained on activity measurements of slag and baghouse dust; consequently, no activity balance was made.

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[17] A subsequent publication reported that the composition of the slag was 72% SiO_2, 13% Al_2O_3, 4.5% Na_2O, 5.0% K_2O and 0.7% CaO (WOR93).

MEN90 Menon, S., G. Hernborg and L. Andersson, "Melting of Low-Level Contaminated Steels," Studsvik AB Sweden, 1990.

Studsvik AB in Sweden has a three-ton induction melting furnace where low-level radioactive scrap is remelted. Based on the melting of 33.61 tons of carbon steel, the weight of ingots was 32.27 tons, the weight of slag was 1.32 tons and the weight of dust was 0.019 tons. No Cs-137 was measured in the ingots and the activity levels in the slag were also below the measurement threshold for the detection equipment. Dust contained the following nuclides:

- Co-60 ..... 1,300 Bq/kg
- Zn-65 ... 14,400 Bq/kg
- Cs-137 21,800 Bq/kg

Studsvik also reported on the results of two stainless steel melts weighing a total of 5,409 kg. The weight of slag in melt 92 was 1.1% of the total and in melt 93 it was 0.5%. The weight of dust from the combined melts was 2.49 kg. Activity measurements are listed in the following table.

Specific Activities of Ingots and Slags (Bq/kg)

Melt No.	Co-58/Co-60	Mn-54	Cs-134/Cs-137	Ag-110m	Sb-125	Zn-65
92 (ingot)	1350	8.2		54	29	34
92 (slag)	720	73	2320	30		
93 (ingot)	3440				50	
93 (slag)	207	10	1493			
Baghouse dust	264/31,200	146	1,125/134,650	37,450	670	52,250

MER93 Meraikib, M., "Manganese Distribution Between a Slag and a Bath of Molten Sponge Iron and Scrap," in *ISIJ International,* Vol. 33, No. 3, pp. 352-360, 1993.

The manganese distribution ratio is given by the expression:

$$\eta_{Mn} = \frac{(Mn)}{[Mn]}$$

$$= a_{[O]}\, f_{[Mn]} \exp\left(\frac{27005}{T} - 7.2324\right)$$

for temperature range from 1,550 to 1,670°C). This equation is based on 80 metal samples from melts in a 70-ton electric arc furnace, and reflects Meraikib's finding a limited influence of slag basicity on the Mn distribution ratio. A different expression, explicitly including the influence of basicity was presented in Section E.5.14.

Extensive thermodynamic calculations are included.

NAK93 Nakamura, H., and K. Fujiki, "Radioactive Metal Melting Test at Japan Atomic Energy Research Institute," 1993.

Air melting was accomplished in a high frequency (1,000 Hz) induction furnace of 500 kg capacity. Researchers studied the effects of melting temperature, slag basicity and type of steel (ASTM-A335 and SUS 304) on partitioning using radioactive tracers: Mn-54, Co-60, Sr-85, Zn-65 and Cs-137. The slag basicity (CaO/SiO_2) was 1 for A335 and 3 for SUS 304. Five radioactive tracer heats (three ASTM-A335 and two SUS 304) and six JPDR decommissioning heats were produced. The average material balance was 99.5%, with the maximum difference being 3%. Material distribution was: 95% ingot, 2-3% slag, 0.1% dust, 1-2% other (metal on tundish and metal splash). The melt temperature was 1,873 K. Results from one of the three A335 tracer tests are as follows:

- Mn-54: recovery 98%, about 7% of which was in slag, balance in *ingot* (approximate Mn content of other three ingots was 90%)
- Co-60: 99.5% recovery, all in ingot
- Zn-65: 90.7% recovery, about 14% of which was in exhaust gas, 1% in slag and balance in ingot
- Sr-85: 72.7% recovery, 100% in slag
- Cs-137: 77% recovery, 50% of which was in slag and 50% in exhaust gas

The other four tracer tests showed similar tendencies.

The melt was held at temperature for about 20 minutes after tracers were added before casting the ingot. Tracers were not present in initial melt charge, but rather were added after melting was completed and the desired temperature of 1,873 K was reached. Exhaust gas analyses were based on sampling about 0.04% of total exhausted volume.

OST94 Ostrovski, O., "Remelting of Scrap Containing Tungsten and Nickel in the Electric Arc Furnace," in *Steel Research*, vol. 65, No. 10, pp 429-432, 1994.

This paper discusses partitioning of W between slag and melt during melting of W-bearing steel scrap in a 25-ton electric arc furnace with slags of varying basicity. Melting under strongly oxidizing conditions (30 min. oxygen blow) and high CaO/SiO2 ratio resulted in 94% of W in slag, 4% in metal and 2% lost. Thermodynamic equations for calculating the partition ratio are provided.

PFL85 Pflugard, K., C. R. Gomer and M. Sappok, "Treatment of Steel Waste Coming From Decommisioning of Nuclear Installations by Melting," in *Proceedings of the International Nuclear Reactor Decommissioning Planning Conference*, NUREG/CP-0068, p. 349-371, Bethesda, MD, July 16-18, 1985.

Sappok described nine melts totaling 24 Mg (plus starting blocks, *i.e.*, furnace heel) in 10-ton and 20-ton induction furnaces. Mass balance: 28,000 kg steel, 800 kg slag, 20 kg furnace lining, and 64 kg cyclone and baghouse dust. Co-60 and Cs-137 distributions were:

Co-60: 97% in steel, 1.5% in slag, 1.5% in cyclone and baghouse
Cs-137: 90% in slag, 1% in furnace lining, (5% in baghouse tubes and dust).

Activities accounted for: Co-60–96%; Cs-137–73%.

No discussion of slagging practices or melting practices and temperatures was included.

Gomer used a 500 kg high frequency induction furnace, a 5-Mg EAF and a 3-Mg BOF (no results reported). Non-quantitative tests from two 5-Mg arc furnace melts showed that all the Co-60 was reported in the melt; quantities in slag and fume were below detection limits. Traces of Am-241 were found in slag when melting contaminated heat exchanger tubing in the arc furnace. The results of three quantitative tests of Cs in 5-ton electric arc furnaces and one in 500 kg induction furnace are listed in Table E-5 of the present report.

Gomer notes that Cs stays in slag in an induction furnace and can be made to stay largely in slag in an arc furnace but conditions "may not be fully practical in production furnaces." No information on melting and slagging practice is included.

SAP90 Sappok, M., *et al.*, "Melting of Radioactive Metal Scrap from Nuclear Installations," in *Decommissioning of Nuclear Installations,* Elsevier Applied Science, pp. 482-493, 1990.

Melting to date has totaled 2000 tons of steel (steel presumed from Pflugard *et al.*, but not so stated in report) in a 20-ton induction furnace. (A new dedicated facility with a 3.2-ton medium frequency induction furnace had recently been completed but no radioactive scrap had yet been melted in the new equipment). When melting Zn-plated metal, Zn is "found in the filter dust." Typical mass balance: 98.6% metal, 1.2% slag and 0.2% filter dust.

For the melting period May 17, 1985: Ce-144 all in slag, Zn-65 all in off-gas, Mn-54 distributed between slag and off-gas, Cs-134/137 distributed between slag and off-gas, Co-60 mostly in melt but some in slag and some in off-gas (Co-60 is only the activity detected in the melt).

For the melting period September 27-28, 1985: Mn-54 distributed between slag and off-gas; Zn-65 all in off-gas; Eu-154 all in slag; Ag-110m distributed among metal, slag and off-gas; Cs-134/137 distributed between slag and off-gas; Co-60 distributed among melt, slag and off-gas, but mostly in the melt.

For the melting period January 1, 1986 – March 14, 1986 (200 tons): Cs-134/137 distributed between slag and off-gas; Mn-54 distributed between slag and off-gas; Zn-65 distributed among slag, metal and off-gas; Ag-110m distributed among slag, metal and off-gas, but mostly in metal; Co-60 distributed among slag, metal and off-gas, but retained mostly in metal.

No discussion of slagging or melting practice was included.

SCH90 Schuster, E., and E. W. Haas, "Behavior of Difficult to Measure Radionuclides in the Melting of Steel," Siemens Aktiengesellschaft Unterehmensbereich KWU, 1990.

Laboratory melts were made using a Nernst-Tammann high-temperature furnace with temperatures to 1,700°C and a 3 to 5 kg melt size. Melt additions included: 1) electro-deposited Co-60, Fe-55 and Am -241 on steel disks, 2) carbonate or hydroxide precipitates or elemental C on SiO_2 filters, 3) direct insertion of U and UO_2. The melts were allowed to solidify in the carborundum tube crucible. About 60% to 80% of the slag was recovered when melting St37-2 steel under Ar + 10% H_2. Results are presented in the following table:

Distribution of Radionuclides Following Laboratory Melts

Sample Location	Percentage of Nuclide in Each Medium			
	Co-60	Fe-55	Ni-63	C-14
Ingot	108	70	≈ 82	91
Slag	0.2	n.d.	0.04	0.4
Aerosol Filter	0.2	n.d.	0.06	< 0.001

In a test for Sr distribution where slag-forming oxides CaO, SiO_2 and Al_2O_3 were added, the Sr-85 distribution was: surface layer of ingot - *ca.* 80%, slag - 6.3%, ingot - 0.5%, aerosol filter - 0.02%. In a test with Am-241, the isotope distribution was: ingot - 1%, slag - 110% and aerosol filter - 0.05%. In tests with UO_2, *when slag formers were added,* the uranium concentration in the ingot was reduced from 330 μg per g to 5 μg per g.

STA61 Starkey, R. H., *et al.*, "Health Aspects of the Commercial Melting of Radium Contaminated Ferrous Metal Scrap," in *Industrial Hygiene Journal*, pp. 489-493, December 1961.

Melting of 40 tons of radium-contaminated steel scrap blended with 20 tons of uranium-contaminated steel scrap in an electric arc furnace is discussed. Based on eight heats, the average concentration of radium in steel ingots was $<9 \times 10^{-11}$ g of Ra per g of steel and radium content of slag was 1.47×10^{-9} g Ra per g of slag. No information on melting and slagging conditions was provided.

STU84a Stubbles, J. R., "Tonnage Maximization of Electric Arc Furnace Steel Production: The Role of Chemistry in Optimizing Electric Furnace Productivity - Part V," in *Iron and Steelmaking*, vol. 11, No. 6, pp. 50-51, 1984.

Stubbles notes that recovery (from scrap) of Cb, B, Ti, Zr, V, Al and Si in steel is zero and recovery of Mo, Ni, Sn and Cu is 100%. Pb, Zn and Sb are volatilized. Cr and Mn are distributed between slag and metal based on the degree of slag oxidation (the "FeO" level). Cr recovery ranges from about 30% to 50% and Mn recovery from about 10% to 25%. No supporting information is provided for these recovery values. According to Stubbles, lead from babbitts, batteries, *etc.* melts and quickly sinks to the furnace bottom, often penetrating the refractory lining. However, when leaded scrap is added to liquid steel, the lead will go into solution and boil off like zinc, exiting with the fume.

STU84b Stubbles, J. R., "Tonnage Maximization of Electric Arc Furnace Steel Production: The Role of Chemistry in Optimizing Electric Furnace Productivity - Part VII," in *Iron and Steelmaking*, vol. 11, No. 8, pp. 46-49, 1984.

Stubbles cites the following charge to produce one ton of liquid steel:

metals	2,100 lb
flux	40 lb
gunning material (high MgO)	10 lb
charge carbon	10 lb

In this example, the initial slag volume is 100 lb per ton. Most input sulfur remains in metal and is extremely difficult to transfer to slag. The theoretical sulfur distribution $\frac{(S)}{[S]}$ rarely exceeds 8 in EAF's. Working down sulfur during melting requires constant removal of high basicity slag plus agitation.

One reason for adding excess carbon above desired final level is to use decarb oxygen from a lance to promote slag/metal reactions and help boil out hydrogen. Hydrogen levels on the order of 1 ppm can be obtained after a 15-minute carbon boil where the rate of carbon removal is 1%/hr. If the C removal rate is 0.1%/hr, the comparable hydrogen level is about 5 ppm (based on an initial level of 9 ppm).

APPENDIX E-2

COMPOSITION OF BAGHOUSE DUST

Various studies have reported measurements of the composition of baghouse dust. Results of measurements reviewed in this study are reported here.

Babcock and Wilcox Company (KAE74) provided the baghouse dust composition at its No. 3 EAF melt shop at Koppel, PA. The melt shop included one 50-ton, one 75-ton and three 100-ton furnaces used for the production of carbon, alloy and stainless steels. The dust composition (in wt%) was:

Fe_2O_3	52.7
CaO	13.6
Al_2O_3	0.9
SiO_2	0.9
MgO	12.6
Mn_2O_3	0.6
ZnO	6.3
NiO	0.1
Cr_2O_3	0.6
CuO	0.1
Loss on ignition at 1100°C	6.8
Balance	4.6

The average dust collection was 12 lb per ton of steel melted. More recently, dust collection has been increasing, reaching a level of 26 lb per ton of carbon steel melting capacity in 1985 and 30 lb per ton of carbon steel melting capacity in 1992 (ADL93).

Arthur D. Little (ADL) prepared a survey on EAF dust generation for the Electric Power Research Institute in 1993 based on 52 shops which melted carbon steel (ADL93). ADL estimated that about 600,000 tons of dust were generated in 1992 from U.S. carbon steel operations. The dust composition (in wt%) was:

Fe	28.5
Zn	19.
Cd	< 0.01
Pb	2.1
Cr	0.39
CaO + MgO	10.7

The high levels of Zn in the dust are the result of large amounts of galvanized steel in the furnace charge. According to ADL the disposition of the baghouse dust in 1992 was:

• Disposal to landfill	11.2%
• Shipped to fertilizer	2.3%
• Shipped to Zn recovery	86.5%
• Miscellaneous, delisted	0.1%

Lehigh University conducted a study on EAF dust for the Department of Commerce in 1982 (LEH82). Dust composition from stainless steel and carbon steel melts is shown in the table below.

Composition of Baghouse Dust (wt%)

Component	Stainless Steel Dust	Carbon Steel Dust
Fe	31.7	35.1
Zn	1.0	15.4
Cd	0.16	0.028
Pb	1.1	1.5
Cr	10.2	0.38
CaO	3.1	4.8

SAIC (McK95) described the composition of EAF dust taken from an earlier work by Brough and Carter (BRO72). The dust composition (in wt%) as quoted by Brough and Carter and interpreted by SAIC is:

Fe_2O_3 .. 52.5
ZnO 16.3
CaO 14.4
MnO 4.4
SiO_2 ..... 2.6
MgO 1.9
Na_2O 1.5
Cl_2 .... 1.2
Other ... 5.2

Based on the original source, Cl_2 should be Cl^- and 4.4% of "Other" is ignition loss. The dust was a by-product of melting low alloy carbon steels.

REFERENCES: APPENDIX E-2

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APPENDIX F

DISTRIBUTION OF CONTAMINANTS DURING MELTING OF CAST IRON

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DISTRIBUTION OF CONTAMINANTS DURING MELTING OF CAST IRON

This appendix discusses the expected partitioning of contaminant elements during the production of cast iron. The approach taken here is to use the information developed for the partitioning of elements during the melting of carbon steel in electric arc furnaces (see Appendix E) and by analogy predict their expected behavior during the production of cast iron. To the extent possible, the deductive process takes into account differences in melting and slagging practice. This discussion should be viewed as a supplement to the information developed in Appendix E. Many of the same references are used as information sources and the detailed thermodynamic discussion is not repeated here.

In order to assess radiation exposures to products made of potentially contaminated cast iron, it is necessary to estimate the partitioning to cast iron of the elements listed in Table 6-3 of the main report. The present discussion of partitioning during the production of cast iron is therefore limited to these elements.

F.1 BACKGROUND

Cast iron is an alloy of iron and carbon (*ca.* 2 to 4.4 wt%) which also typically contains Si, Mn, S and P. The high carbon content of the alloy results in a hard, brittle product which not amenable to metalworking (as is steel) and hence the alloy is cast into the desired end use form. As noted by the United States Steel Corporation, now USX, (UNI51):

> Castings are of innumerable kinds and uses, roughly grouped as chilled-iron castings, gray-iron castings, alloyed-iron casting and malleable castings. In general, castings are made by mixing and melting together different grades of pig iron; different grades of pig iron and foundry scrap; different grades of pig iron, foundry scrap and steel scrap; different grades of pig iron, foundry scrap, steel scrap, and ferroalloys and other metals.

Chemical compositions of typical cast irons are presented in Table F-1 (EPA95).

Cast iron is typically melted in a cupola furnace, an electric arc furnace, an electric induction furnace or an air (reverberatory) furnace. A flow diagram for a typical iron foundry is shown in Figure F-1 (EPA95). The cupola is similar to a small blast furnace where the iron ore in the charge is replaced by pig iron and steel scrap. As described in UNI51:

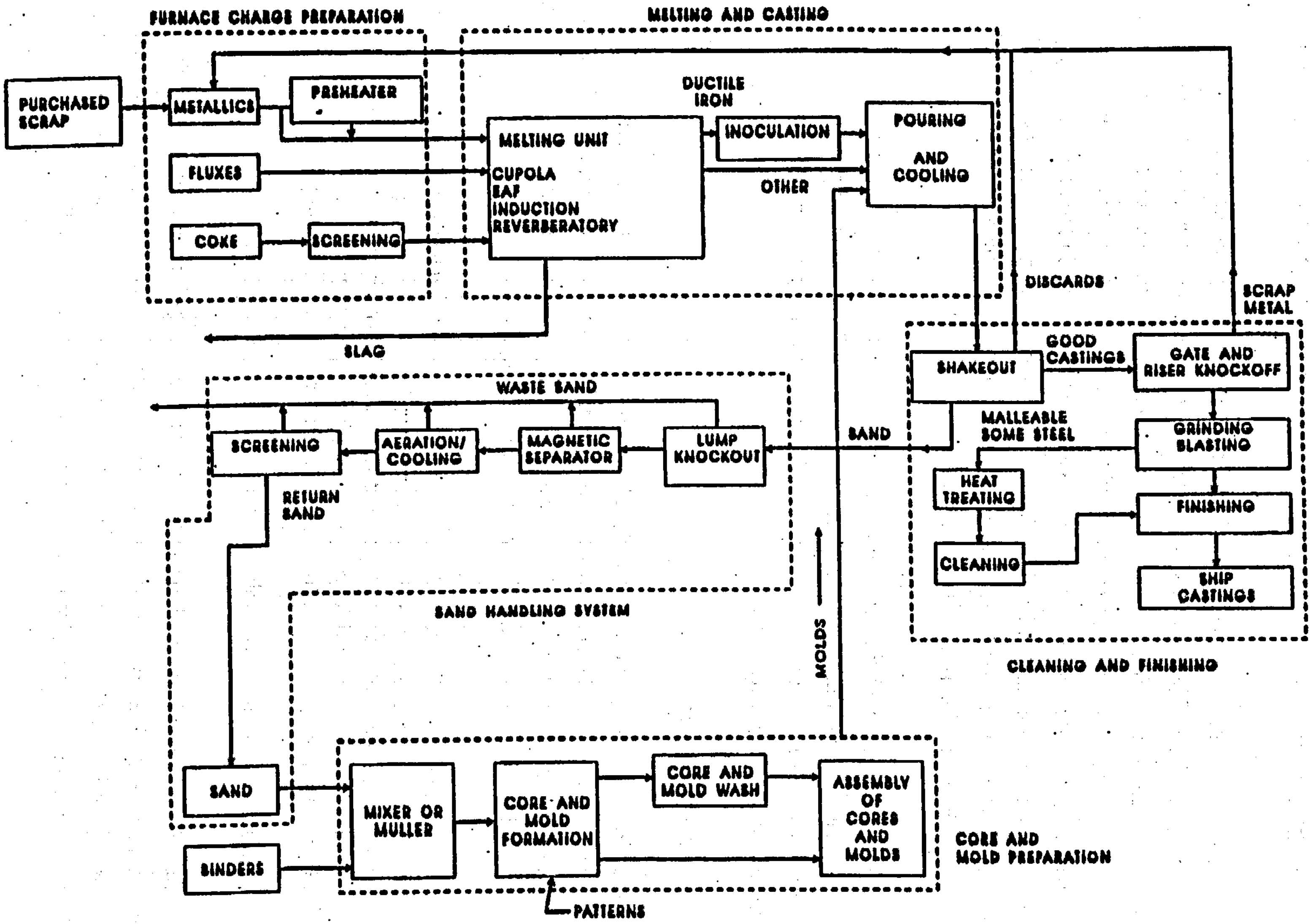

Figure F-1. Flow Diagram of a Typical Cast Iron Foundry (from EPA95)

The charge is composed of coke, steel scrap and pig iron in alternate layers of metal and coke. Sufficient limestone is added to flux the ash from the coke and form the slag. The ratio of coke to metallics varies depending on the melting point of the metallic charge. Ordinarily, the coke will be about 8 to 10 per cent of the weight of the metallic charge. It is kept as low as possible for the sake of economy and to exclude sulphur and some phosphorus absorption by the metal.

During melting, the coke burns as air is introduced at a 10 to 20 ounce pressure through the tuyeres. This melts the charge as some of the manganese combines with the sulfur forming manganese sulphide which goes into the slag. Some manganese and silicon are oxidized by the blast and the loss is proportional to the amount initially present. Carbon may be increased or reduced depending on the initial amount present in the metallic charge. It may be increased by absorption from the coke or oxidized by the blast. Phosphorus is little affected but sulfur is absorbed from the coke. Prior to casting the slag is removed from the slag-off hole which is located just below the tuyeres. The molten metal is then tapped through a hole located at the bottom level of the furnace. The depth between these two tapping holes and the inside diameter of the furnace governs the capacity of the cupola.

Table F-1. Chemical Composition of Ferrous Castings (wt%)

Element	Gray Iron	Malleable Iron (as white iron)	Ductile Iron	Steel Scrap[a]
Carbon	2.0 - 4.0	1.8 - 3.6	3.0 - 4.0	0.18 - 0.23
Manganese	0.40 - 1.0	0.25 - 0.80	0.5 - 0.8	0.60 - 0.90
Phosphorus	0.05 - 1.0	0.06 - 0.18	<0.15	≤ 0.40
Silicon	1.0 - 3.0	0.5 - 1.9	1.4 - 2.0	—
Sulfur	0.05 - 0.25	0.06 - 0.20	<0.12	≤ 0.05

[a] Nominal composition of a low carbon steel (*e.g.*, SAE 1020)

The melt temperatures used in producing cast irons are lower than those used in steel making. The melting point of pure iron is 1,532°C, while steel making temperatures are typically about 1,600°C. Furthermore, carbon depresses the melting point of iron: the melting point of an iron alloy containing 3.56% C and 2.40% S is 1,250°C, while one containing 4.40% C and 0.6% Si has a melting point of 1,088°C (UNI51).

Fluxing agents added to the furnace charge to promote slag formation include carbonates (*e.g.*, limestone and dolomite), fluorides (*e.g.*, fluorspar), and carbides (*e.g.*, calcium carbide)

(EPA95). Obviously, the furnace environment during the production of cast iron is more highly reducing than that in typical steel melting.

Emissions from the cast iron melting furnaces include particulate matter, carbon monoxide, sulfur dioxide, and small quantities of chlorides and fluorides. These emissions are from incomplete combustion of carbon additives, oxidation of sulfur in coke (for cupola melting), flux additions, and dirt and scale in the scrap charge (EPA95). Melting of ductile iron requires the addition of inoculants such as magnesium in the final stages of melting. The Mg addition to the molten bath results in a violent reaction and the production of MgO particulates and metallic fumes. Most of these emissions are captured by the emission control system and routed to the baghouse, where the fumes are cooled and filtered. Cupolas are also equipped with an afterburner in the furnace stack to oxidize the carbon monoxide and burn any organics.

In 1993, U.S. shipments of iron and steel castings were (BUR95a):

Ductile iron castings	3,740,000 tons
Gray iron castings	9,110,000 tons
Malleable iron castings	292,000 tons
Total	13,140,000 tons

Scrap consumption by iron foundries and miscellaneous users in that year is summarized below (BUR95b):

Electric arc furnace	4,630,000 tons
Cupola furnace	9,230,000 tons
Air furnaces and other	66,000 tons
Total	13,920,000 tons

In addition, 744,000 tons of pig iron and 6,000 tons of direct reduced iron were consumed by the iron foundries. The total metal consumption was 14,670,000 tons, which is about 12% greater than cast iron shipments. This difference may be due to scrap generation or inventory changes. From a recycling perspective, a significant observation is that cast iron contains about 95% scrap metal.

In 1989, about half of all iron castings were used by automotive and truck manufacturing companies and half of all ductile iron castings were used in pressure pipe and fittings (EPA95).

F.2 MATERIAL BALANCE

Using the results of several studies, EPA95 has compiled emission factors for uncontrolled emissions from two types of gray iron foundries:

Cupola furnace	13.8 lb/ton metal
Electric arc furnace	12.0 lb/ton metal

F.2.1 Cupola Furnaces

Based on a 1980 EPA-sponsored environmental assessment of the iron casting industry, Baldwin (BAL80) reported that a typical cupola producing a medium strength cast iron from a cold charge would utilize the following materials (as a percentage of iron input):

scrap steel	48%
foundry returns (i.e., foundry home scrap)	52%
ferrosilicon	1.1%
ferromanganese	0.2%
coke	14%
limestone	3%
melting loss	2%

Baldwin also documented the quantities of material produced for three foundries: a malleable iron foundry using a induction furnace, a ductile iron foundry using a cupola, and a gray and ductile iron foundry using a cupola for primary melting which duplexes into induction furnaces. The amounts of by-products are listed in Table F-2.

Table F-2. Amounts of By-Products from Various Foundries

By-Product	Amount Generated (lb per ton of metal melted)		
	Malleable Iron	Ductile Iron	Gray and Ductile Iron
Slag	34.5	173	130
Dust Collector Discharge	7.19		78.6

F.2.2 Electric Arc Furnaces

According to a study conducted for EPA, a typical charge for an electric arc furnace (EAF) includes (JEF86):

- 50% - 60% scrap iron
- 37% - 45% scrap steel
- 0.5%- 1.1% silicon
- 1.3%- 1.7% carbon raisers[1]

Arc furnaces for cast iron melting range from 500 pounds to 65 tons capacity, 25 tons being a common size (BAL80). According to JEF86, 94% to 98% of the EAF charge is recovered as iron.

F.2.3 Chemistry Adjustments

As noted in Section F.2.1 and F.2.2, the furnace charge typically contains about 45% steel scrap. If this scrap were similar to that listed in the last column of Table F-1, then, to achieve the cast iron chemistries indicated in that table, it would be necessary to add C, P, S, Si, and possibly Mn to the furnace charge.

Production of ductile iron requires making additions to the melt which alter the shape of the graphite particles in the cast iron from flakes to a spheroidal form. Typically, the melt is inoculated with magnesium just before pouring to produce the ductile iron. Much of the magnesium boils off in the process. Sometimes Ba, Ca, Ce, Nd, Pr, Sr and Zr are also added as inoculants (BAL80). To reduce the costs of adding magnesium in larger ductile iron production operations, the melt is desulfurized before magnesium is added. This is frequently done by adding CaC_2 (BAL80).

F.3 PARTITIONING BASED ON REDUCTION OF FeO IN SLAG

As discussed in Section E.4 of Appendix E, an indication of contaminant partitioning between the melt and the slag can be obtained by calculating the free energy change for the reaction

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[1] Carbon raisers are additives introduced into the bath to increase the carbon content of the cast iron, if required.

$$M + \left(\frac{y}{x}\right)FeO = \left(\frac{y}{x}\right)Fe + \left(\frac{1}{x}\right)M_xO_y \qquad \text{(F-1)}$$

where M is the pure component rather than the solute dissolved in the melt and FeO and M_xO_y are slag components. The standard free energies of reaction of various contaminants with FeO at 1,873 K, a typical temperature for the production of carbon steel in an EAF, were presented in Table E-2. Recalculation of these values for a temperature of 1,573 K, which is typical for cast iron production, indicates no substantive changes from the previous conclusions regarding which elements are expected to concentrate in the slag and which are expected to concentrate in the melt. The assumed 300 K temperature difference between steel melting and cast-iron melting produces small changes in the free energies of Equation F-1 but no significant shifts in the expected equilibria. The free energies of reaction at 1,573 K are listed in Table F-3.

F.4 ADJUSTMENTS TO HENRY'S LAW FOR DILUTE SOLUTIONS

Partition ratios presented in Table E-1 for carbon steel were also recalculated for a furnace temperature of 1,573 K. While slight changes in partitioning ratios were obtained at the lower temperature, no significant shifts in equilibria resulted. An example is the comparable partition ratios for cobalt and uranium, which are shown in Table F-4.

Calculations of partition ratios at 1,573 K are summarized in Table F-5. Values of $\gamma°$ were calculated using temperature dependent values of the free energy change for transference of the pure substance to a dilute solution in liquid iron. All values were obtained from SIG74 except Ce which was taken from JAP88.

F.5 OBSERVED PARTITIONING DURING METAL MELTING

F.5.1 General Observations

Because of concerns that tramp elements might be accumulating in cast irons from contaminants in steel scrap and affecting casting behavior, the U.S. Bureau of Mines conducted an extensive study over a period of more than five years to evaluate the impurities in cast iron (NAF90). While this study does not specifically address partitioning, the results can provide confirmation of inferred partitioning. Samples were obtained from 28 ductile iron foundries and 52 gray iron foundries at various times over the course of the study. The distribution of foundries by geographical location, furnace type and product is shown in Table F-7.

Table F-3. Standard Free Energy of Reaction of Various Contaminants with FeO at 1,573 K

Element	Oxide	ΔF° (Kcal)	Comments
$Ac_{(l)}$	Ac_2O_3	-121	Ac should partition to slag
$Am_{(l)}$	Am_2O_3	-105	Am should partition to slag
$Ba_{(l)}$	BaO	-59.6	Ba should partition to slag
$Cs_{(l)}$	Cs_2O		Cs_2O unstable at 1,573 K, Cs should vaporize from melt, some Cs may react with slag components
$Np_{(l)}$	NpO_2	-104	Np should partition to slag
$Pa_{(l)}$	PaO_2	-100	Pa should partition to slag
$Pu_{(l)}$	Pu_2O_3	-89.1	Pu should partition to slag
$Ra_{(g)}$	RaO	-55.0	Ra should partition to slag
$Ru_{(s)}$	RuO_4		Ru should remain in melt
$Sb_{(g)}$	Sb_2O_3		Sb will not react with FeO, some may vaporize from melt
$Sr_{(g)}$	SrO	-65.8	Sr should partition to slag, but low boiling point could cause some vaporization
$Tc_{(s)}$	TcO_2		Tc will not react with FeO, should remain in melt
$Th_{(s)}$	ThO_2	-147	Th should partition to slag
$Y_{(l)}$	Y_2O_3	-104	Y should partition to slag
$Zn_{(g)}$	ZnO		Zn will not react with FeO, Zn should vaporize from melt

Table F-4. Partition Ratios for 2 Elements at Typical Iron- and Steel-Making Temperatures

Element	Partition Ratio (N_{mo}/wt% M)	
	1,573 K	1,873 K
Cobalt	1.0E-4	4.8E-5
Uranium	1.4E+8	8.9E+7

Table F-5. Partition Ratios at 1,573 K for Various Elements Dissolved in Iron and Slag

M	Oxide	γ°_{M}	$\Delta F^{\circ}_{\ell MO}$ (kcal/mole)[a]	Partition Ratio (N_{MO}/wt%M)
$Ag_{(l)}$	Ag_2O	546	+16.5	1.06E-03[b,c]
$Al_{(l)}$	Al_2O_3	0.013	-280	2.63E+05[b]
$Ca_{(g)}$	CaO	1330	-118	1.15E+10
$Ce_{(l)}$	Ce_2O_3	0.26	-302	1.79E+07[b]
$Co_{(l)}$	CoO	1.08	-25.0	1.00E-04
$Cr_{(s)}$	Cr_2O_3	1.45	-111	1.86E-03[b]
$Cu_{(l)}$	Cu_2O	12.9	-14.0	2.56E-03[b]
$Mn_{(l)}$	MnO	1.36	-64.3	5.24E+00
$Mo_{(s)}$	MoO_3	2.60	-95.3	3.49E-06
$Nb_{(s)}$	Nb_2O_5	1.79	-298	1.22E+05[b]
$Ni_{(l)}$	NiO	0.51	-25.1	4.98E-05
$Pb_{(l)}$	PbO	11900	-17.8	4.56E-02
$Si_{(l)}$	SiO_2	2.7E-4	-143	4.00E+01
$Sn_{(l)}$	SnO_2	3.44	-61.7	3.70E-05
$Ti_{(s)}$	TiO_2	0.035	-159	2.22E+05
$U_{(l)}$	UO_2	0.014	-193	1.44E+08
$V_{(s)}$	V_2O_5	0.078	-228	9.93E+00[b]
$W_{(s)}$	WO_3	1.73	-110	6.56E-05
$Zr_{(s)}$	ZrO_2	0.029	-191	4.52E+08

[a] $\Delta F^{\circ}_{\ell FeO} = -38.1$ Kcal/mole

[b] PR = $N^{1/2}$/wt% M

[c] Ag will not react with FeO, Ag_2O unstable at 1,573K

With limited exceptions, Ce, Nb, Pb and Sb were not found at the limits of detection listed below (wt%) for the 23 calendar quarters over which sampling was conducted:

- Ce 0.02 - 0.1 (wt%)
- Nb 0.01 - 0.05 (wt%)
- Pb 0.005 - 0.2 (wt%)
- Sb 0.02 - 0.1 (wt%)

Lead levels above the lower detection limit were detected in four quarters as shown in Table F-6.

Table F-6. Lead Levels at Two Different Type Foundries

Quarter	Pb Above Detection Limits (wt%)	
	Ductile Iron	Gray Iron
1	0.005–0.007	< 0.005–0.007
2	< 0.005–0.008	< 0.005–0.010
3		< 0.005–0.006
20		< 0.005–0.007

Average analyses for other elements of interest are included in Table F-8.

F.5.2 Antimony

Thermodynamic calculations based on Equation F-1 indicate that antimony will not partition to the slag. Experimental work by Kalcioglu and Lynch (KAL91) showed that when antimony is added to carbon-saturated iron at 1,723 K and allowed to react with an acidic slag (basicity ratio = 0.666), the resulting partition ratios were those listed in Table F-9.

Based on these values for L_{Sb} and an assumed slag to metal mass of 0.05, the quantities of Sb in the slag are insignificant (*i.e.*, <1%). Antimony recoveries ranged from 47 to 71% for these four tests, the losses being presumably due to vaporization.

NAS93 cites the following equation for the activity of Sb in carbon-saturated iron:

$$\log \gamma^{\circ}_{Sb} = -\frac{6623}{T} + 5$$

This yields a value for γ° of 6.2 at 1,573 K, which, when combined into the Henry's Law relationship, indicates that the partition ratio, $\frac{(N_{Sb_2O_3})^{1/2}}{[\text{wt\% Sb}]}$ is 2.6 x 10^{-5}, supporting the view that Sb partitions strongly to the melt. Although, as noted in Section F.5.1, no Sb was found in cast iron samples at the lower limit of detection (0.02 wt%), this does not necessarily vitiate the thermodynamic partitioning argument. Sb may not be present in the feed materials at the detection limit. Although some antimony may vaporize from the melt, insufficient evidence is available to quantify this possibility. To avoid possibly underestimating exposures to cast iron products potentially contaminated with Sb, Sb is assumed to remain in the melt.

Table F-7. Distribution of Foundries in Bureau of Mines Tramp Element Study

Geographic Zone	Ductile Iron						Gray Iron					
	Furnace Type			Furnace Size[a]			Furnace Type			Furnace Size[a]		
	Cupola	Electric	Induction	A	B	C	Cupola	Electric	Induction	A	B	C
Northeast	1	0	2	1	1	1	6	0	2	3	5	0
Great Lakes	5	0	2	1	2	4	12	0	2	4	7	3
Southeast	1	1	3	3	1	1	4	0	3	3	2	2
Upper Midwest	4	1	3	0	8	0	11	1	4	0	12	4
West	1	0	4	5	0	0	3	1	3	5	1	1

[a] A, < 1,000 tons per month; B, 1,000 to 8,000 tons per month; C, >8,000 tons per month

Table F-8. Average Concentrations of Tramp Elements in Cast Iron (wt%)

Zone	Ductile Iron					Gray Iron				
	Co	Mn	Mo	Ni	Zn	Co	Mn	Mo	Ni	Zn
Northeast	0.008	0.378	0.020	0.067	0.003	0.009	0.726	0.025	0.073	0.002
Great Lakes	0.007	0.405	0.022	0.117	0.003	0.010	0.703	0.051	0.192	0.002
Southeast	0.009	0.453	0.017	0.171	0.004	0.010	0.675	0.030	0.142	0.003
Upper Midwest	0.008	0.409	0.024	0.257	0.002	0.009	0.701	0.040	0.107	0.002
West	0.012	0.415	0.025	0.186	0.005	0.009	0.670	0.040	0.086	0.002

Table F-9. Distribution of Sb Between Slag and Metal

[wt%Sb]	L_{Sb}[a]
0.45	0.067
0.87	0.022
1.03	0.020
1.06	0.018

[a] L_{Sb} = (wt%Sb)/[wt%Sb]

F.5.3 Carbon

As was noted in Sections F.2.1 - F.2.3, carbon is added to the furnace charge to achieve the levels desired in the finished product (*e.g.* 1.8% to 4.0% C). During the melting process, some of the carbon in the scrap steel may be oxidized and removed from the system as CO; however, there is a net addition of carbon to the melt, rather than a net removal. Since it is impossible to predict how much carbon is removed from the scrap steel and later replaced with carbon from other charge materials, it is conservative to assume that all the carbon in the scrap remains in the cast iron.

F.5.4 Cerium

Cerium is sometimes used as an inoculant in ductile irons (BAL80); consequently, small amounts must remain in the melt, in spite of the fact that thermodynamic calculations suggest that Ce partitions strongly to the slag. In addition, as noted in Section F.5.1, Ce was not found in cast iron at the limits of detection in samples from 28 ductile iron foundries. Given this conflicting information, the most likely situation is that minute amounts of Ce will remain in the cast iron. However, no evidence has been uncovered which suggest that the amount of Ce remaining in the melt is greater than 0.5% of the total.[2]

F.5.5 Cesium

All Cs is expected to partition between the slag and the baghouse dust. None is expected to remain in the melt (HAR90).

F.5.6 Iron

Some iron is expected to be oxidized and to transfer to the slag. However, no detailed composition data have been located in this study to permit quantification of this expected partitioning. Therefore, it is conservatively assumed that no iron is transferred to the slag.

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[2] Partition ratios in the present analysis are calculated to the nearest 1%. Thus, any partition ratio less than 0.5% is equated to zero.

F.5.7 Lead

Based on thermodynamic equilibrium calculations, Pb is expected to remain in the melt. However, Pb has very limited solubility in liquid iron. Furthermore, it has a vapor pressure of 0.01 atm at 1,408 K (DAR53) and 0.05 atm at 1,462 K (PER84). At the limits of detection, lead is seldom found in cast iron (see Section F.5.1).

Lead has been detected in leachates from baghouse dust collected by cupola emission control systems. Leachate levels based on the EP toxicity test ranged from about 10 to about 220 mg/L (KUN90). Since it is not possible to quantitatively relate these leachate results to contaminant levels in the dust, one can only reach the qualitative conclusion that some Pb volatilizes from the cast iron melt and is collected in the baghouse.

The combined evidence indicates that, for the purposes of the present analysis, lead can be assumed to completely vaporize from the melt.

F.5.8 Manganese

Based on thermodynamic calculations which assume that $\gamma^{\circ}_{Mn} = 2.6$, the partition ratio of manganese between slag and iron is calculated to be about 5 at 1,573 K (see Table F-5), which suggests that significant amounts of Mn will be present in both the slag and the melt. Meraikib (MER93) determined that during steelmaking, the distribution of Mn between the slag and the melt could be described by the equation

$$\eta_{Mn} = \frac{(Mn)}{[Mn]}$$

$$= a_{[O]} f_{[Mn]} \exp\left(\frac{27530}{T} - 0.0629\,B - 7.3952\right)$$

where:

(Mn) = concentration of Mn in slag (wt%)
$[Mn]$ = concentration of Mn in melt (wt%)
$a_{[O]}$ = activity of oxygen in melt
$f_{[Mn]}$ = activity coefficient for [Mn]
T = absolute temperature (K)
B = slag basicity

Although there are risks in extrapolating this equation to cast iron melting, the calculation was undertaken in the absence of better information. Partition ratios at two different partial pressures of CO were estimated, assuming $T = 1{,}573$ K, $B = 0.63$, $f_{[Mn]} = 0.95$, and 130 lb of slag generated per ton of metal melted. These values are listed in Table F-10.

Table F-10. Partition Ratios of Manganese at Different Partial Pressures of CO[3]

P_{CO} (atm)	η_{Mn}	Partition Ratio (see text) (mass in slag/mass in metal)
1	0.45	0.03
0.1	0.045	0.003

F.5.9 Niobium

On the basis of thermodynamic calculations, niobium is expected to partition primarily to the slag. However, according to Harvey (HAR90), Nb can be retained in steel under reducing conditions. The expected reaction is

$$2\underline{Nb} + 6\underline{O} + Fe = FeO{\cdot}Nb_2O_5$$

where the elements on the left side of the equation are melt constituents and the compound on the right is a slag constituent. The equilibrium constant for the reaction is

$$K_1 = \frac{a_{FeO\cdot Nb_2O_5}}{a_{Fe}\, a_{\underline{Nb}}^2\, a_{\underline{O}}^6} = 2.4\cdot 10^{10} \qquad (T = 1{,}873K)$$

Assuming that $\frac{a_{FeO\cdot Nb_2O_5}}{a_{Fe}} = 1$, values of $a_{\underline{Nb}}$ corresponding to two assumed values of $a_{\underline{O}}$ can be calculated, as listed below:

---

[3] The oxygen activity is calculated using free energy values for C and O dissolved in iron (JAP88) and the CO free energy of formation from GLA57. The calculated values are in close agreement with information presented in ENG93 (p. 67).

$a_{\underline{O}}$	$a_{\underline{Nb}}$
1	6.5E-6
0.01	6.5

The value of K_{1573}, the equilibrium constant at 1,573 K, is not available; however, based on the values of the free energies of formation of Nb_2O_5 at 1,573 K and 1,873 K, it is expected that $K_{1573} > K_{1873}$. Thus, a highly reducing environment ($a_{\underline{O}} \ll 1$) would be required to retain Nb in the melt at the lower temperature.

As noted in Section F.5.1, Nb is not detected in cast iron at the detection limit, which indicates that either there are no significant quantities of Nb in steel scrap or the typical melting conditions are not sufficiently reducing to cause Nb to be retained in the melt.

F.5.10 Zinc

Under steelmaking conditions, Zn is expected, from a free energy perspective, not to partition to the slag and, because of its high vapor pressure, to vaporize from the melt to a large extent. Cast-iron melting temperatures, though lower, are still well above the normal boiling point of Zn (1,180 K).

Based on information presented by Perrot *et al.*, the solubility of Zn at 1,573 K is expected to be about 140 ppm when the partial pressure of Zn is 10^{-2} atm. Silicon in the cast iron will tend to increase the Zn solubility while Mn will have the opposite effect (PER92). As noted in Section F.5.1, from 20 to 50 ppm of Zn are typically found in cast iron, which suggests that it is unrealistic to assume that 100% of the Zn volatilizes and collects in the baghouse. Assume, for example, that a furnace charge contains 45% steel scrap and 50% cast iron scrap, and that both the cast iron scrap and the product contains 30 ppm Zn, as listed in Table F-8. If the steel scrap contains less than 0.67 wt% Zn, then 1% or more of the Zn would remain in the melt (see Note 2). (KOR94).

According to Koros, typical galvanized scrap contains about 2% Zn (KOR94). The same author reported that, in 1992, 35% of the scrap classified as No. 1 bundles and busheling is galvanized steel. Other grades of scrap likely to contain significant quantities of galvanized steel include shredded scrap and No. 2 bundles (FEN96). For 1993, No. 1 bundles, No. 1 busheling, shredded, and No. 2 bundles accounted for 46% of the carbon steel scrap used in iron foundries (BUR95b). Using the above information, it can be estimated that about 2% of the Zn will remain in the cast iron and the balance will be transferred to the baghouse dust, based on the following calculation:

$$P_{Fe}^{Zn} = \frac{C_{Fe}^{Zn}}{f_{Fe}^{Fe'} C_{Fe'}^{Zn} + f_{Fe}^{s} f_{s}^{g'} f_{g'}^{g} C_{g}^{Zn}}$$

P_{Fe}^{Zn} = partition fraction of zinc in cast iron
= 0.0185

C_{Fe}^{Zn} = mass fraction of zinc in cast iron product
= 3×10^{-5}

$f_{Fe}^{Fe'}$ = mass ratio of cast iron scrap:cast iron product
= 0.45

$C_{Fe'}^{Zn}$ = mass fraction of zinc in cast iron scrap
= 3×10^{-5}

f_{Fe}^{s} = mass ratio of steel scrap:cast iron product
= 0.5

$f_{s}^{g'}$ = fraction of galvanized-steel-bearing scrap sources in steel scrap
= 0.46

$f_{s}^{g'}$ = fraction of galvanized steel in galvanized-steel-bearing scrap sources
= 0.35

C_{g}^{Zn} = mass fraction of zinc in galvanized steel
= 0.02

F.6 PARTITIONING SUMMARY

F.6.1 Elements Which Partition to the Melt

It is assumed that 1% of the total melt will be transported from the furnace and collected in the baghouse. This is approximately the geometric mean of the values for two types of foundries listed in Table F-2 and is consistent with the values cited in EPA95 (see Section F.2). Based on thermodynamic equilibria, the following elements are expected to partition 99% to the melt and 1% to the baghouse dust: Co, Mo, Ni, Ru, and Tc.

Free energy calculations also suggest that silver partitions to the melt but, for EAF melting of carbon steel, this information was tempered by the facts that silver has a significant vapor pressure at steelmaking temperatures (10^{-2} atm at 1,816 K) and some work on stainless steel melting done at Studsvik (MEN90) had shown Ag in the baghouse dust. However, the vapor pressure of Ag is at least an order of magnitude lower at temperatures used in cast iron melting (*e.g.*, 10^{-3} atm at 1,607 K)(DAR53). Consequently, in cast iron, silver is assumed to partition 99% to the melt and 1% to the baghouse dust.

Although there is reason to suspect that some Nb might be found in the melt under highly reducing conditions, no evidence was uncovered to support that supposition.

For reasons discussed in Section F.3.3 above, carbon and antimony are expected to remain in the melt except for small quantities of dust transferred to the baghouse (*i.e.*, 1%).

Manganese is predicted to remain primarily in the melt. It is expected that no more than about 2% of the manganese will partition to the slag.

Most of the zinc is expected to volatilize and be collected in the baghouse. Only about 2% is assumed to remain in the melt.

Table F-11 lists the partition ratios of elements which are expected to show significant (*i.e.*, at least 1%) partitioning to the melt.

F.6.2 Elements Which Partition to Slag

For those elements which are strong oxide formers and are expected to partition to the slag, the assumption is made here that 5% of the slag will be transported to the baghouse as dust. This is the same assumption as made for melting carbon steel in electric arc furnaces. Based on this assumption, thermodynamic equilibrium calculations at 1,573 K and chemical analogies, the following elements are expected to partition 95% to the slag and 5% to the baghouse dust: Ac, Am, Ce, Cm[4], Eu[4], Nb, Np, Pa, Pm[4], Pu, Ra, Sr, Th, and U.

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[4] Since thermodynamic data were not available for these elements, partitioning was assumed to be analogous to similar elements in the rare-earth and auctioned series in the periodic table.

Table F-11. Proposed Partitioning of Metals Which Remain in the Melt

Element	Distribution (%)		
	Melt	Slag	Baghouse
Ag	99		1
C	99		1
Co	99		1
Fe	99		1
Mn	97	2	1
Mo	99		1
Ni	99		1
Ru	99		1
Sb	99		1
Tc	99		1
Zn	2		98

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REVIEW DRAFT

TECHNICAL SUPPORT DOCUMENT

EVALUATION OF THE POTENTIAL FOR RECYCLING OF SCRAP METALS FROM NUCLEAR FACILITIES

VOLUME 3 OF 3: APPENDICES G-L

Prepared by:

S. Cohen & Associates, Inc.
1355 Beverly Road
McLean, Virginia 22101

Under

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Prepared for:

U.S. Environmental Protection Agency
Office of Radiation and Indoor Air
401 M Street, S.W.
Washington, D.C. 20460

Martin Offutt
Work Assignment Manager

July 15, 1997

VOLUME 3

APPENDICES G-L

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APPENDIX G

DILUTION OF SCRAP METAL FROM NUCLEAR FACILITIES

PREFACE

This Appendix describes the development of dilution factors for use in the exposure assessment of the RMEI to materials and products associated with the recycling of potentially contaminated carbon steel scrap. This analysis was updated to utilize the most recent estimates of the anticipated yield of potentially contaminated, recyclable carbon steel scrap generated by the dismantlement of commercial, light-water nuclear reactors, which appears in Appendix A of this report. The discussion of dilution factors in Section 5.2.1 of Chapter 5, however, relies on earlier results of the reactor dismantlement analysis. Chapter 5 estimates that 13% of the steel in the scrap yard and 11% at the steel mill would be potentially contaminated, assuming that 17,000 tons of potentially contaminated scrap would be recycled at a single facility in the peak year. The more recent analysis presented in this Appendix cites a value of 22,500 tons, which, combined with other assumptions which remain unchanged, would result in a contaminated scrap fraction of 17% in the scrap yard and 15% at the steel mill, an increase of about 30%. Most of the normalized doses and risks listed in Table 7-1 in Chapter 7 and in Appendices J and K would increase in the same proportion. The exceptions are the normalized doses and risks calculated for end-users of finished products, scenarios in which dilution factors were not applied.

The recycling analysis presented in this Appendix represents but one approach to estimating dilution factors. This analysis will be re-evaluated along with alternate plausible scenarios during subsequent refinements of the RMEI exposure assessment. The results of that assessment will be presented in a forthcoming revised version of this report.

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DILUTION OF SCRAP METAL FROM NUCLEAR FACILITIES

G.1 INTRODUCTION

Chapter 5 discusses the operations and scenarios used to assess the radiation exposures of the RMEI during the recycling of potentially contaminated scrap metal. Each operation exposes the individual to materials or products generated during a certain stage of the recycling process. It is unlikely that for an entire year,[1] any scrap processor would be exclusively supplied with scrap resulting from the dismantling of components that had been potentially exposed to radioactive contamination. To determine the largest fraction of scrap that would be potentially contaminated, the anticipated release of scrap metal by various generator sites nationwide were matched to the scrap processing capacities of nearby steel mills. This appendix presents a discussion of that analysis.

In its 1996 survey of EAF furnaces in the United States, *Iron and Steelmaker* (I&SM) listed 213 furnaces with a combined nominal capacity of 57,850,000 tons per year (ISM96). The largest capacity furnace in their survey was a 370-ton furnace with a nominal capacity of 950,000 tons per year and the smallest was a 10-ton furnace with an annual capacity of 4,000 tons. The average annual capacity from the I&SM survey is 272,000 tons per furnace. EAF steel production in 1995 was 40,619,000 tons (AIS95), which suggests that the industry was running at about 70% of capacity in that year.

One important factor in developing worker exposure scenarios is the number of furnaces at a site. If there are multiple furnaces at a site, the worker exposure will be related to the total steel tonnage produced at the site rather than the tonnage produced by a single furnace. Recognizing the importance of these and other factors, one can make some estimates as to how operating conditions may alter worker exposure when melting scrap metal from nuclear facilities. First, we will consider an average exposure case and subsequently a reasonable maximum exposure case.

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[1] The potential radiological impacts on the RMEI are assessed during the year of peak exposure.

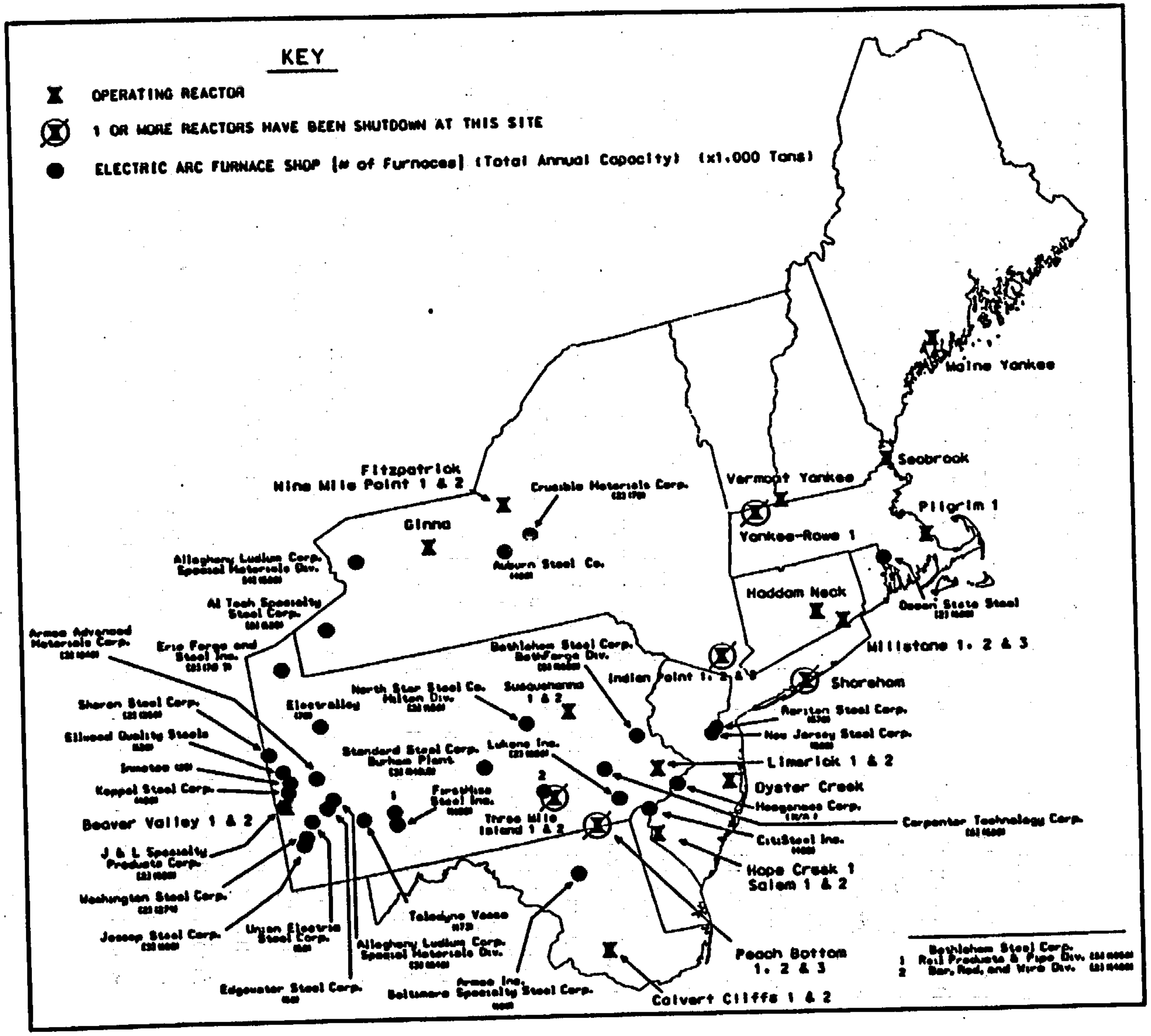

Figure G-1. Electric Arc Furnace Shops in NRC Region I (Northeast)

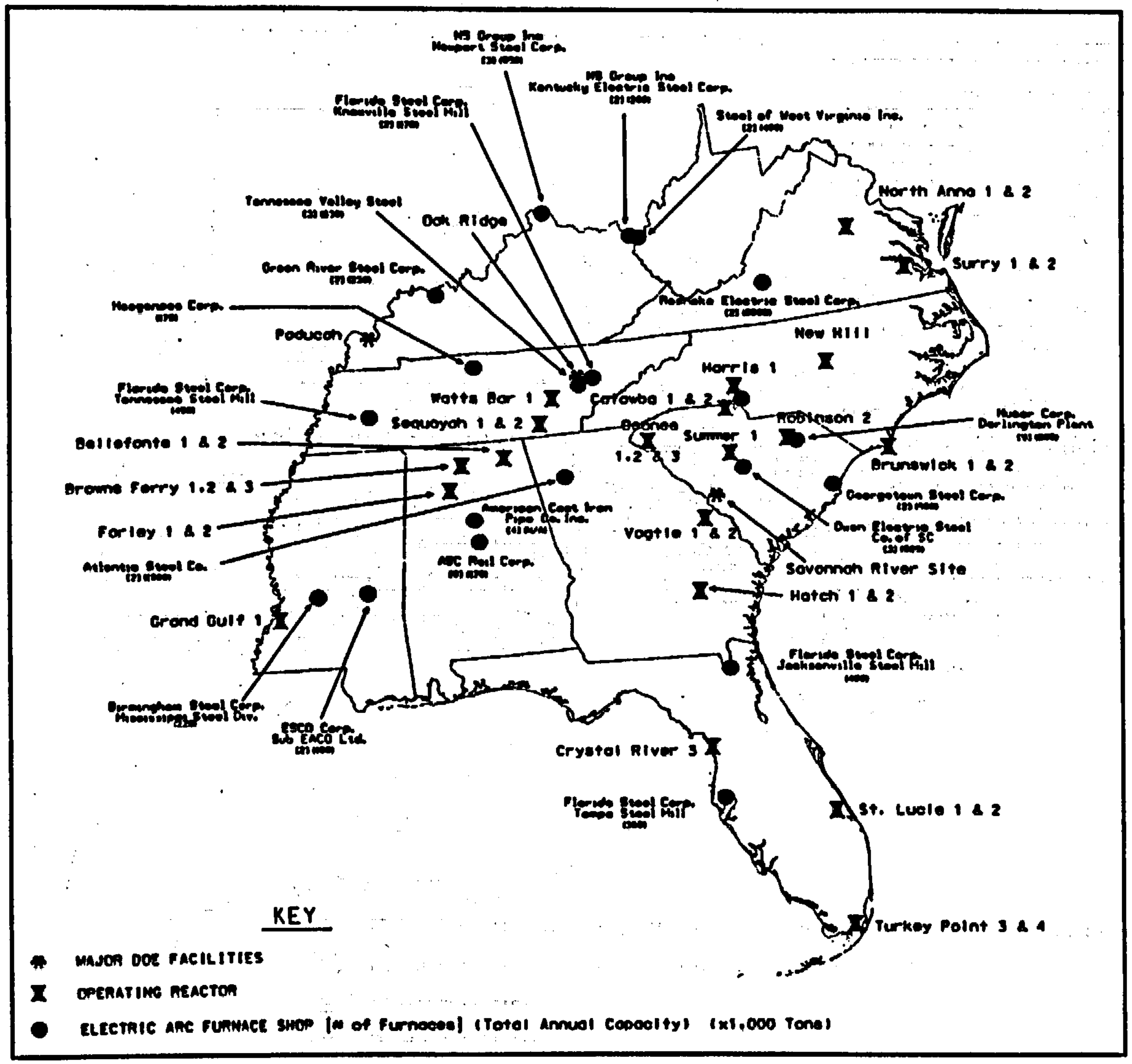

Figure G-2. Electric Arc Furnace Shops and Nuclear Facilities in NRC Region II (Southeast)

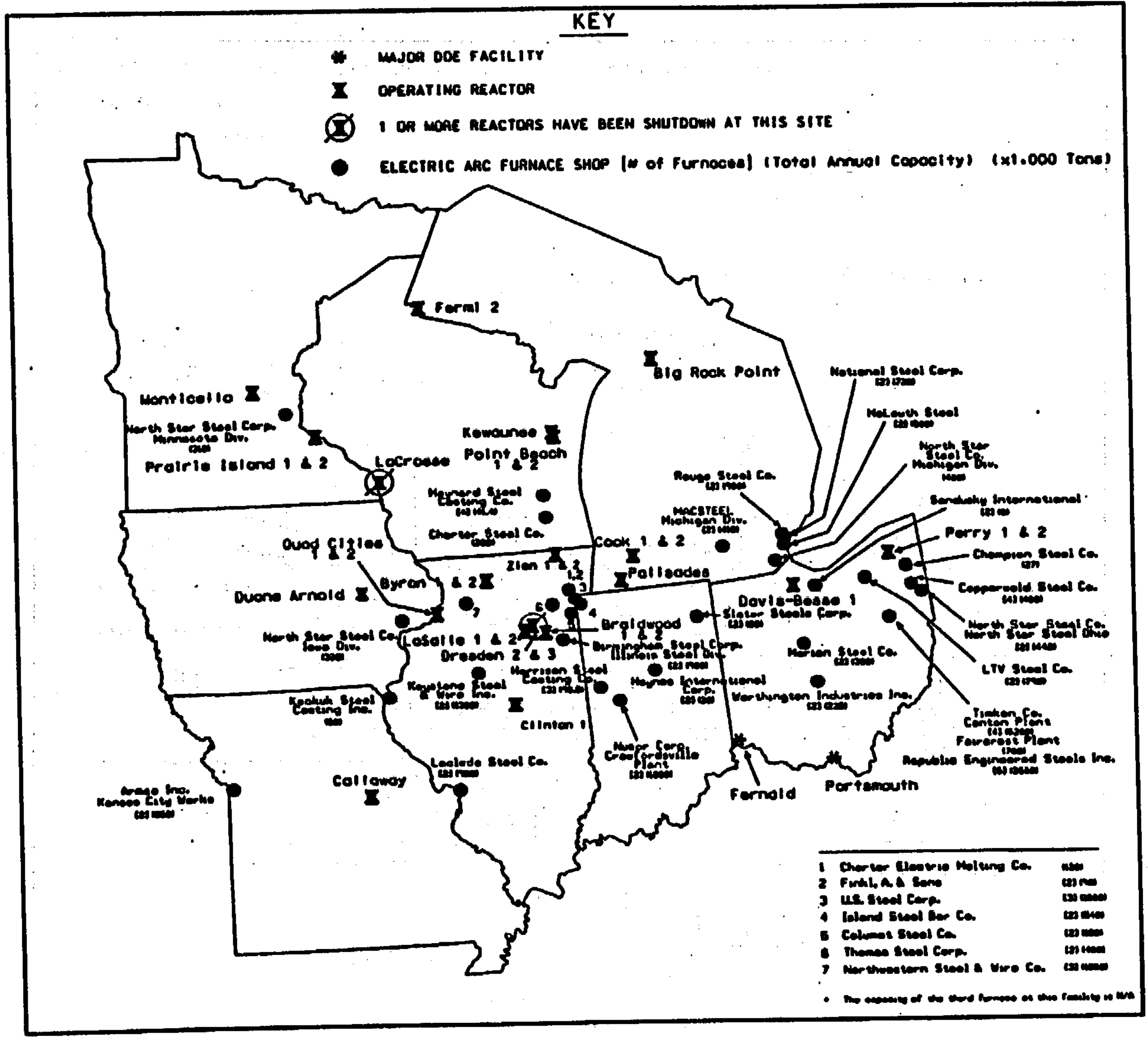

Figure G-3. Electric Arc Furnace Shops and Nuclear Facilities in NRC Region III (North Central)

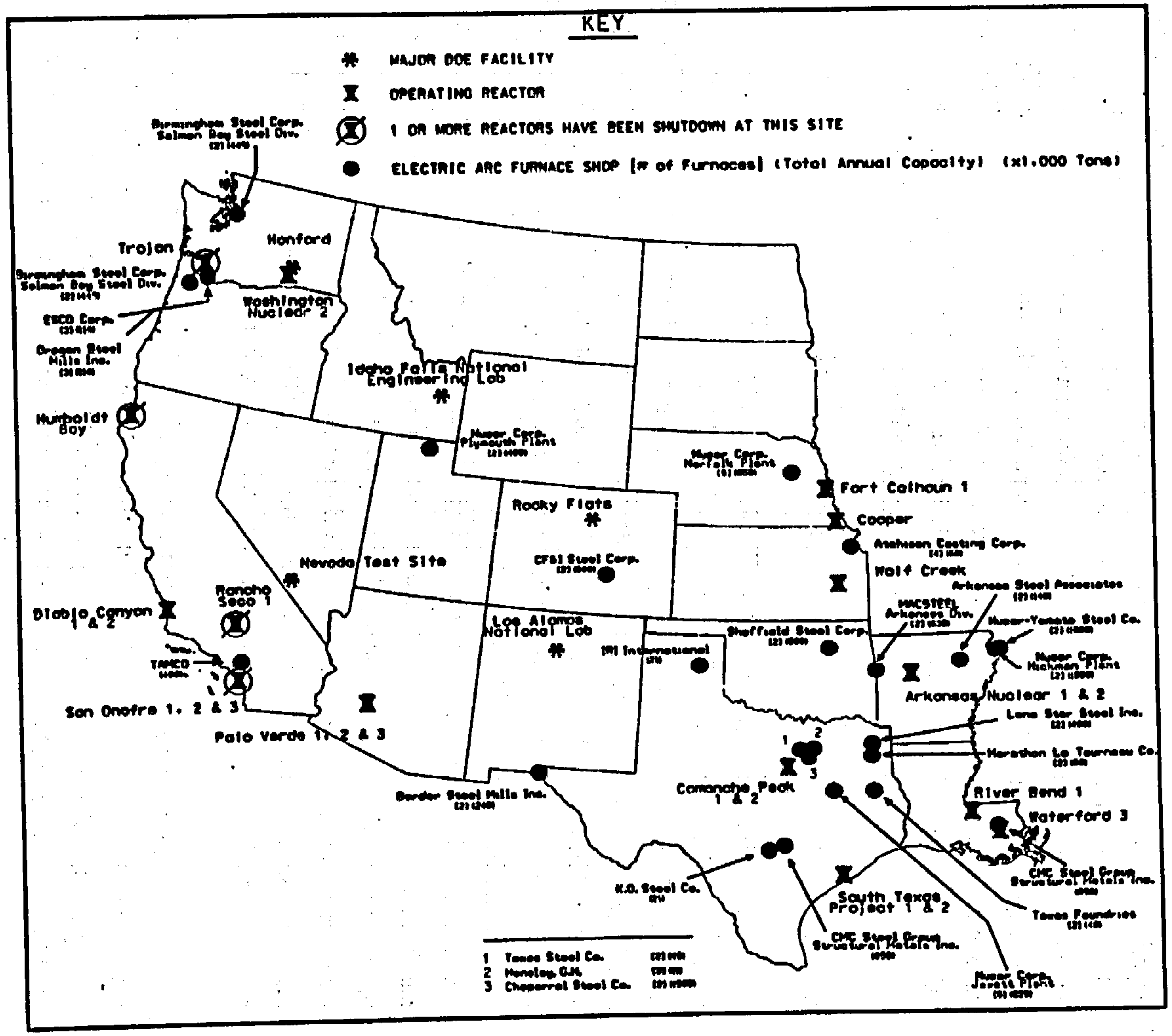

Figure G-4. Electric Arc Furnace Shops and Nuclear Facilities in NRC Region IV (West)

G.2 AVERAGE CASE

According to Table A4-4 of Appendix A, an estimated four million tons of carbon steel from decommissioned commercial nuclear power plants will potentially be available for recycling. Assuming a 40-year operating life of a reactor and a 10-year delay between shutdown and release of scrap metal for recycling, the process would continue through 2053. If the first reactor were to be decommissioned in the year 2000, an average of 73,500 tons of carbon steel scrap would be generated each year during this period. If all of this tonnage were shipped as scrap to a single "average" EAF, it would represent 27% of the annual capacity *of that furnace only*. If it were evenly distributed among all the furnaces in the U.S., the carbon steel scrap from decommissioned nuclear power plants would represent 0.13% of total EAF capacity.

G.3 REASONABLE MAXIMUM EXPOSURE (RME) CASE

For administrative purposes, the NRC has divided the 48 contiguous states into four regions, which are depicted in Figures G-1 to G-4. Superimposed on these maps are the locations of EAFs, as well as the locations of nuclear power plants and major DOE facilities that constitute present and future sources of potentially contaminated scrap metal. These maps show that both EAFs and nuclear facilities are broadly distributed across the country. A cursory examination reveals that, with two exceptions, each state that is host to a nuclear facility also has one or more EAF shops or is adjacent to a state that has such shops.[2] Since transportation costs would be a major factor in determining which EAF shop receives the scrap from a given nuclear facility, the geographical distribution of nuclear facilities and scrap melters should lead to the scrap being distributed among many EAFs. However, the simultaneous shutdown of two or more reactors in the same vicinity could lead to a potential release of a relative large amount of scrap at a single location for a brief period of time. A few hypothetical examples of such releases, and their consequences, are discussed in this section.

To develop the reasonable maximum exposure (RME) case, we will first assume that scrap metal tends to move the shortest possible distance to minimize transportation costs. For example, when the five nuclear power plants in southern California (San Onofre 1, 2 and 3, and Diablo Canyon 1 and 2) are dismantled, we assume that the carbon steel scrap would be shipped

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[2] The exceptions are Maine and New Hampshire. The nuclear plants in these states are nevertheless closer to the nearest EAFs than are some of the nuclear facilities in the West. The scales of the maps, which are different for the Northeast and Western regions, may give a different visual impression.

to TAMCO, near Riverside, CA, for melting. Based on the projected year of shutdown and a 10-year delay, scrap from these five reactors would be available for recycling between 2002 and 2036. Two of these five reactors, San Onofre 2 and 3, are anticipated to be shut down in 2013 (see Addendum 1 to Appendix A). Although decommissioning of a reactor can take several years (SMI78), for the purpose of a conservative analysis, it is assumed that all the recyclable scrap metal would be released in a single year. According to Table A4-4, the decomissioning of a reference PWR would result in approximately 36,000 tons of carbon steel scrap being potentially available for recycling. Applying the scaling factors[3] that reflect the power ratings of these reactors, and assuming a ten-year delay between shutdown and release of scrap metal, we find that approximately 76,000 tons would be available in 2023. This is about 19% of the nominal annual TAMCO capacity of 400,000 tons *for that year only*. By the same logic, the other three reactors, each scheduled to be shut down in a different year, would use 10% or less of the TAMCO capacity in any one year.

Not all the carbon steel from a commercial reactor consists of the potentially contaminated, recyclable metal that is the subject of this analysis. Some of the scrap generated during decommissioning would never have been exposed to radioactive contamination, while other metal would have neutron activation products throughout its volume and would thus not be a candidate for free release. Table A5-4 indicates that a maximum of 3,311 metric tons of carbon steel from a reference 1,000 MWe PWR and 6,754 metric tons of carbon steel from a reference 1,000 MWe BWR would be potentially suitable for recycling. Again applying the appropriate scaling factors and converting to English units, we find that only 7,700 tons of potentially contaminated scrap from San Onofre 2 and 3 could be available for recycling. Such scrap would constitute less than 2% of TAMCO's nominal annual capacity.

In this hypothetical scenario, any stainless steel available for recycle would have to be shipped elsewhere since TAMCO is a carbon steel shop.

The peak years for reactor shutdowns would be 2013 and 2014, with 13 reactors reaching the end of their 40-year operating lives during each of these two years.[3] Again assuming a ten-year delay between shutdown and release of scrap metal, 423,000 tons would be released in 2023 and 406,000 tons in 2024. Four of the 13 reactors due to shut down in 2013 are in northern

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[3] See Addendum 1 to Appendix A.

Illinois—all four are owned by Commonwealth Edison. The dismantling of these four reactor is expected to generate 132,000 tons of carbon steel scrap. This would represent 88% of the capacity of a smaller melt shop such as Calumet Steel Co. in West Chicago, IL, which has a nominal capacity of 150,000 tons per year, but only about 18% of the 750,000-ton annual capacity of the Birmingham Steel Corporation melt shop in nearby Kankakee IL. Only 22,500 tons of this scrap would be potentially contaminated, however, constituting about 15% of a 150,000-ton/year EAF melt shop. No such geographical concentration is projected in 2014, when 13 reactors located in 13 different states are anticipated to shut down.

Since stainless steel melting capacity is less widely distributed geographically, a different scenario is postulated. Using reasoning parallel to that of the carbon steel analysis, we find that about 14,000 tons of potentially contaminated stainless steel would be available for recycling in 2024. If *all* of this stainless steel scrap was processed at a single melt shop, such as Carpenter Technology Corporation's plant in Reading, PA, it would utilize 14% of the plant's nominal capacity for that one year.

G.4 RECOMMENDED APPROACH TO DILUTION

The development of a reasonable maximum exposure case assumes that the scrap steel from the maximum number of reactors decommissioned in any year would all be directed to one of the smaller EAF melt shops in the same state as the decommissioned reactors. Using this approach, it appears that about 15% of the melt shop capacity could be committed to potentially contaminated carbon steel scrap suitable for recycling. *It should be emphasized that this utilization factor would not be sustained in other years at the same shop.*

Factors which could further reduce the quantity of scrap from nuclear facilities melted in a given shop include:

- incompatibility of scrap with product specifications
- incompatibility of large, single-source commitments with other purchasing arrangements
- reluctance to handle such scrap irrespective of actual risks
- scrap buy-back arrangements with customers
- release of scrap from the decommissioning of a reactor over a period of several years

- staggered shutdown of the four Commonwealth Edison reactors to obviate the simultaneous replacement of such a large source of power
- decomissioning of the four reactors would be in tandem rather than simultaneously.

One factor which could possibly increase the use of such scrap by a given recycling facility is the possibility that its price would have to be heavily discounted in comparison to comparable non-nuclear scrap, and that some marginal melt shops might seize the opportunity to purchase cheap scrap for a quick profit.

Based on the information presented here, it is proposed that a reasonable maximum exposure scenario would involve 15% of the EAF shop's capacity being committed to potentially contaminated scrap during the peak year. In any one of five other years during a 54-year period, a maximum of 5% of that shop's capacity would be utilized for the recycling of potentially contaminated scrap. In the other 48 years, no potentially contaminated scrap would be processed.

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APPENDIX H

DETAILED SCENARIO DESCRIPTIONS

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DETAILED SCENARIO DESCRIPTIONS

This Appendix presents detailed discussions of some of the assumptions and parameters used in the analysis of the exposure scenarios presented in Table 5-1. The models for calculating the dose from external exposure and inhalation pathways are described for all applicable cases. The inadvertent ingestion pathway for all applicable scenarios is described in Section 6.3.3, exposure to contaminated drinking water is discussed in Section 6.4.1, and the consumption of food contaminated by residual radioactivity leached from cast iron cooking utensils is discussed in Section 6.4.2. The exposure to fugitive airborne emissions from the furnace is described in Section 6.4.3. Synopses of the CAP-88 analyses are found in Appendix H-2, which is part of the present Appendix.

As can be seen in Table 5-1, the annual exposure duration of most industrial workers is 1,750 hours, which is based on the observation that workers typically spend seven hours of a nominal eight-hour day in close proximity to the potential source of radiation exposure. Exceptions to this assumption are discussed in the following sections of this appendix.

The external exposure rates calculated by the MicroShield™ computer program can be converted to effective dose equivalents for photons incident on an anthropomorphic phantom in one of five geometries:

- anterior-posterior,
- posterior-anterior,
- lateral,
- rotational, and
- isotropic

Since the anterior-posterior (A-P) geometry results in the highest doses and since it is reasonable to believe that workers would spend most of their time facing their work, which is also the source of the external exposure, the A-P orientation was assumed unless otherwise stated.

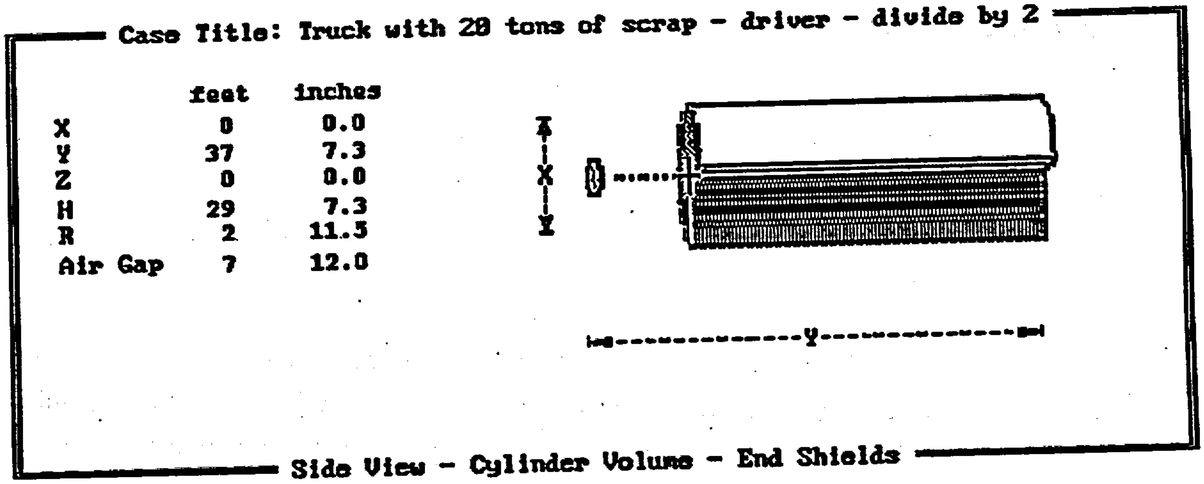
The scenarios are described in the order in which they are listed in Table 5-1, along with the mnemonic by which they are identified in the summary tables of results which appear in Appendix K.

H.1 TRUCK DRIVER TRANSPORTING SCRAP—SCRPDRVR

The truck driver transporting scrap would be exposed to direct penetrating radiation from x- and γ-emitting radionuclides in the load of potentially contaminated scrap. The driver is assumed to spend his/her full time (40 hours per week, 50 weeks per year) in the cab of a truck transporting potentially contaminated scrap. (This is a conservative assumption, since in reality he/she would also be driving the empty truck back for another load.) Since the driver does not come in intimate contact with the material, he/she would not receive any significant internal exposure.

H.1.1 External Exposure

The MicroShield™ computer program was used to calculate normalized dose rates to the scrap truck driver from external exposure. A load of scrap was assumed to weigh 20 tons and to have an average bulk density of 1.57 g/cm³, $\frac{1}{5}$ the density of steel. The load was modeled as a semi-cylinder—the MicroShield™ cylinder geometry was used and the results divided by two. Assuming an aspect ratio of cylinder length to diameter of 5:1, the load was calculated to be approximately 30 feet long and 6 feet wide. The driver was assumed to be located in the cab, 8 feet in front of the load. The MicroShield™ geometry is illustrated in Figure H-1.



Case Title: Truck with 20 tons of scrap - driver - divide by 2

	feet	inches
X	0	0.0
Y	37	7.3
Z	0	0.0
H	29	7.3
R	2	11.5
Air Gap	7	12.0

Side View - Cylinder Volume - End Shields

Figure H-1: Truck Driver MicroShield™ Geometry

H.2 CUTTING SCRAP—SCRAPCUT

The RMEI at the scrap processing facility would be the worker who sections oversized pieces of scrap with a cutting torch. This scrap cutter would be exposed to direct, penetrating radiation from x- and γ-emitting radionuclides in the potentially contaminated scrap, to inhalation of radionuclides that would be volatilized along with the steel during the cutting process, and to inadvertent ingestion of such nuclides in the particulate matter that is generated from the scrap.

H.2.1 External Exposure

As observed during a visit to a large scrap yard, workers spend time in narrow passages—resembling canyons—between mountainous piles of scrap. Since each wall of the canyon constitutes a half-plane, the two walls together can be conservatively modeled as an infinite plane. The doses from external exposures to such an infinite plane can best be calculated by use of the dose coefficients for exposure to soil contaminated to an infinite depth, which are listed in Table III.7 of Federal Guidance Report (FGR) No. 12 (Eckerman 93).

This approach yields a conservative but reasonable estimate of the effective dose equivalent (EDE) in the cases of interest. Since the atomic number of iron, the chief constituent of steel scrap, is considerably higher than that of soil, and since the mass absorption coefficient of energetic photons (x-rays and γ-rays) increases with the atomic number of the absorber, it might at first appear that using soil as a surrogate for steel would understate the absorption and thus significantly overstate the external exposure. However, this did not prove to be the case in the present analysis. As was shown in Table 7-1, the scrap cutter is the RMEI for ten of the radionuclides included in the analysis. In only four of these ten cases does the external exposure pathway make a dominant contribution to the total dose. For these four nuclides, the absorption of a given mass thickness of iron is at most 10% greater than the same mass thickness of soil. Thus, the dose calculated from contaminated soil might be at most 10% greater than the dose from the same nuclides in scrap steel. In light of the variability and uncertainty of other aspects of the analysis, this is not a significant error.

H.2.2 Inhalation of Gaseous or Suspended Radionuclides

According to a scrap yard superintendent (Schiffman 96), a scrap cutter spends up to six hours a day actually cutting scrap—the rest of his time is spent going from one yard location to another or waiting for the scrap to be brought to his location. Since the suspended and vaporized contaminants would be produced by the cutting process, the duration of the cutter's exposure via the inhalation and inadvertent ingestion pathways would be up to 1,500 hours per year. The concentration of dust and vapor in the ambient air is based on an experiment conducted at the Idaho National Engineering Laboratory (Newton 87). Cutting stainless steel pipe with an oxy-acetylene torch in a ventilated enclosure produced average concentrations of respirable particles (0.1 - 10.3 μm AMAD) of 15 mg/m^3. Such a high concentration is unlikely in the worker's breathing zone in an outdoor location. Furthermore, it would be in violation of OSHA PELs, which restrict average *total* dust loading to 15 mg/m^3 and the concentration of respirable particles to 5 mg/m^3. However, since the experiment does indicate the potential for the cutting process to generate high dust concentrations, the average concentration of respirable dust was assumed to be equal to the OSHA PEL of 5 mg/m^3.

H.3 CRANE OPERATOR—OP-CRANE

H.3.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates to the crane operator. The primary source of external exposure would be the charging bucket, which is modeled as a rectangular solid, 30 feet wide, 12 feet high and 10 feet long.[1] Although the bucket has steel walls that are approximately 1 inch thick, the attenuation of radiation by this additional shielding was conservatively neglected. The crane operator was assumed to be 10 meters from the bucket. Advantage was taken of the symmetry of the source to make the best use of the computation time: instead of modeling the entire source volume with the dose point along the central axis, a source with one-half the width and one-half the height, with the dose point along one edge, was modeled. The calculated results were then multiplied by four to

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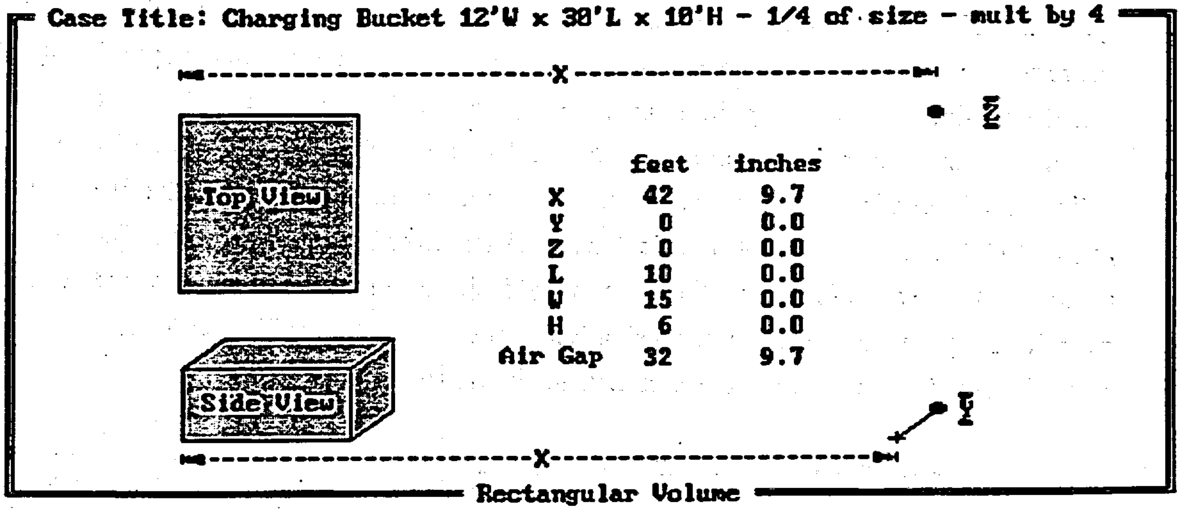
[1] Whenever possible, the designation of rectangular dimensions as length, width and height conforms to the convention of the MicroShield™ program, which always labels the dimension along the X-axis (i.e., towards the dose point) as length. In some cases, as when this dimension is very much smaller than the others, calling it length would be contrary to the conventional understanding of the term.

account for the three missing but identical quadrants. The model geometry is depicted in Figure H-2.



Case Title: Charging Bucket 12'W x 30'L x 10'H - 1/4 of size - mult by 4

	feet	inches
X	42	9.7
Y	0	0.0
Z	0	0.0
L	10	0.0
W	15	0.0
H	6	0.0
Air Gap	32	9.7

Rectangular Volume

Figure H-2: Crane Operator MicroShield™ Geometry

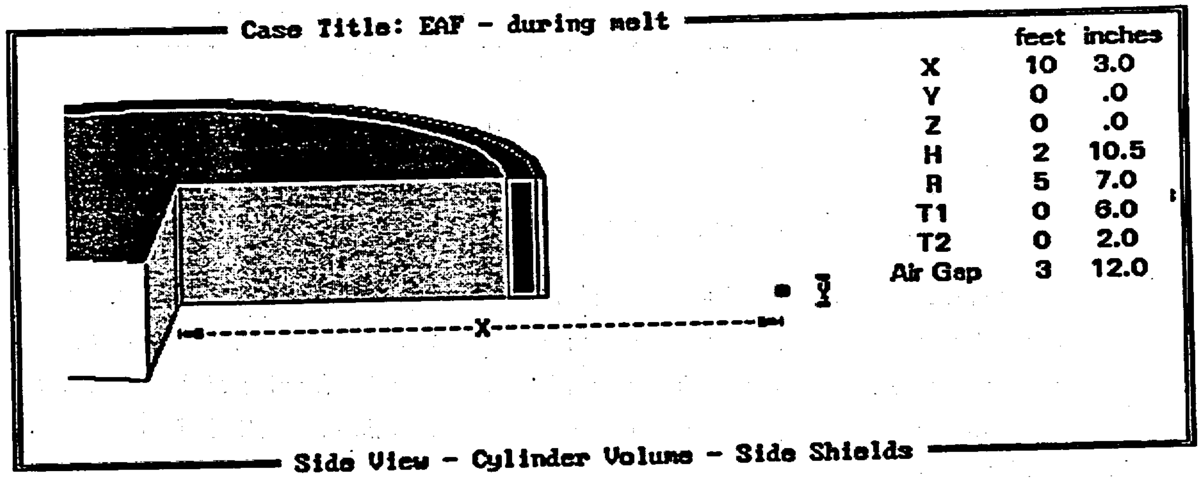
H.3.2 Inhalation of Fugitive Furnace Emissions

The crane operator inhales air containing fugitive furnace emissions. The average dust loading, 1.3 mg/m^3, is modeled on a report of the measured dust concentration at a crane operator's work station at an operating steel mill. The respirable fraction of fugitive furnace emissions in this and other scenarios is taken from the *Compilation of Air Pollutant Emission Factors* (EPA 95).

H.4 EAF FURNACE OPERATOR—FURNACE

H.4.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates to the furnace operator. The electric arc furnace (EAF) of the reference steel mill was based partly on the Calumet Steel Co. facility in Chicago Heights, IL, as described in the May, 1991 *Iron and Steelmaker* (ISM 91). ISM 91 lists a shell diameter of 12.5 feet. Other dimensions were based on the professional experience and judgement of our project team. The furnace was assumed to have a 2-inch thick steel outer shell and a 6-inch thick inner shell of refractory brick, which was modeled as concrete in the MicroShield™ dose calculations. The radiation source in the external exposure assessment was assumed to be a load of potentially contaminated scrap which, prior to melting, has an average bulk density 2 g/cm^3. Advantage was taken of the symmetry of the source to make the best use of the computation time: instead of modeling the entire source volume with the dose point in the plane bisecting the cylinder, a cylinder of one-half the height, with the dose point in the plane containing the base, was modeled. The calculated results were then multiplied by two to account for the missing but identical half of the cylinder. Observations of a furnace operator indicated that his distance from the furnace ranged from 4 to 30 feet. Dose rates were calculated at distances of both 4 and 30 feet. The average normalized dose rates for this worker, assuming his distance from the furnace varied uniformly over this range, were then calculated using Equations 6-5 and 6-6 in Chapter 6. The MicroShield™ geometry for the nearer distance is shown in Figure H-3.


Case Title: EAF - during melt

	feet	inches
X	10	3.0
Y	0	.0
Z	0	.0
H	2	10.5
R	5	7.0
T1	0	6.0
T2	0	2.0
Air Gap	3	12.0

Side View - Cylinder Volume - Side Shields

Figure H-3: Furnace Operator MicroShield™ Geometry

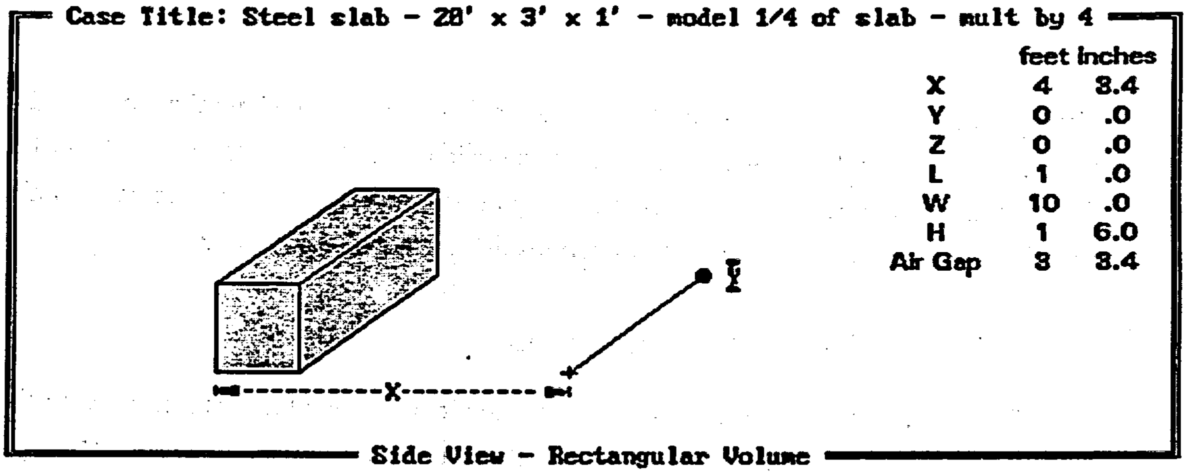
H.4.2 Inhalation of Fugitive Furnace Emissions

The furnace operator would also be exposed to air containing fugitive furnace emissions. The average dust loading, 2.2 mg/m^3, is modeled on a report of the measured dust concentration at a furnace operator's work station at an operating steel mill.

H.5 CONTINUOUS CASTER OPERATOR—OPCASTER

H.5.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates for the operator of the continuous caster. There are two potential sources of external exposure in this scenario: the bloom—a long steel slab that is produced by the caster—and the molten steel in the tundish that feeds the caster. The dimensions of the bloom—20 feet wide, 3 feet high and 1 foot long—are based on conversations with Mr. James Yusko of the Pennsylvania Department of Environmental Resources and on information obtained while touring three steelmaking facilities. Advantage was taken of the symmetry of the source to make the best use of the computation time: instead of modeling the entire source volume with the dose point along the central axis, a source with one-half the width and one-half the height, with the dose point along one edge, was modeled. The calculated results were then multiplied by four to account for the three missing but identical quadrants. The model geometry is depicted in Figure H-4.

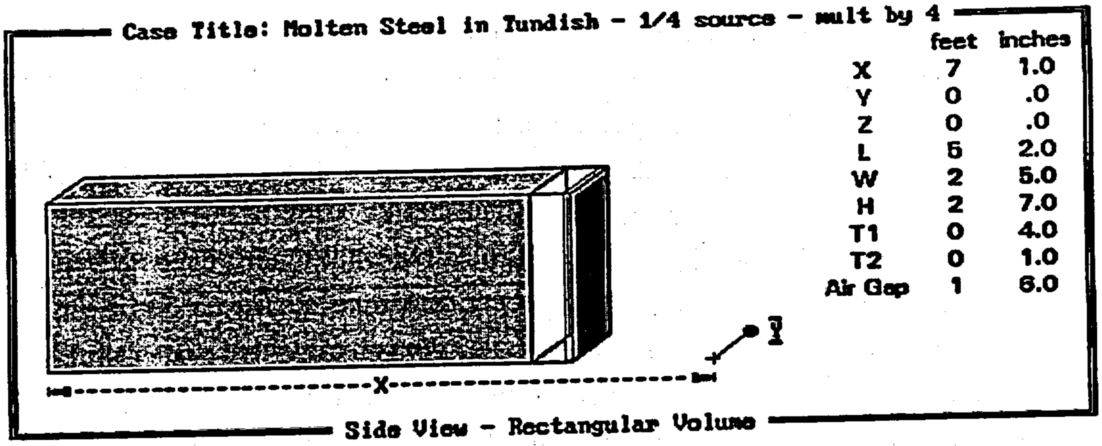

Case Title: Steel slab - 20' x 3' x 1' - model 1/4 of slab - mult by 4

	feet	inches
X	4	8.4
Y	0	.0
Z	0	.0
L	1	.0
W	10	.0
H	1	6.0
Air Gap	8	8.4

Side View - Rectangular Volume

Figure H-4: Continuous Caster Operator MicroShield™ Geometry - Steel Slab

The tundish was modeled as a rectangular solid, 5 feet 2 inches long, 4 feet 10 inches wide and 5 feet 2 inches high, with a 4-inch-thick inner wall of refractory brick and a 1-inch-thick steel outer wall. As in the case of the furnace, concrete, one of the built-in MicroShield materials, was used as a surrogate for the refractory bricks. As before, the source was represented by one quadrant and the results were multiplied by four. The model geometry is shown in Figure H-5.



Case Title: Molten Steel in Tundish - 1/4 source - mult by 4

	feet	inches
X	7	1.0
Y	0	.0
Z	0	.0
L	5	2.0
W	2	5.0
H	2	7.0
T1	0	4.0
T2	0	1.0
Air Gap	1	6.0

Side View - Rectangular Volume

Figure H-5: Continuous Caster Operator MicroShield™ Geometry - Tundish

Observations of a caster operator indicated that his distance from the both the bloom and the tundish ranged from 2 to 15 feet. Dose rates were calculated at distances of both 2 and 15 feet. The average dose rates for this worker, assuming his distance from the furnace varied uniformly over this range, were calculated using Equations 6-5 and 6-6 in Chapter 6.[2]

H.5.2 Inhalation of Fugitive Furnace Emissions

The caster operator would also be exposed to air containing fugitive furnace emissions. The average dust loading, 2.0 mg/m^3, was modeled on a report of the measured dust concentration at a caster operator's work station at an operating steel mill.

H.6 BAGHOUSE MAINTENANCE WORKER—BAGHOUSE

The baghouse maintenance worker was assumed to perform three types of duty during the course of his/her work: maintenance work inside the baghouse, maintenance and monitoring performed outside the baghouse, and routine steel-mill duties not involving the baghouse. His/her annual dose is the sum of the doses received while performing these various tasks.

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[2] The model geometries shown in Figures H-4 and H-5 are for an intermediate distance.

H.6.1 External Exposure

Interior Maintenance

The baghouse consists of 18 modules arranged in two rows, as shown in Figure H-6, below. Each module is 30 feet high; the remaining dimensions are shown in Figure H-6. It contains 72 filters made of Nomex, a material which consists of long-chain polyamides and is chemically similar to nylon. A new filter weighs about 8 pounds, while a used filter, containing residual dust, weighs 18 pounds. Each module is thus modeled as containing 576 pounds of nylon (8 x 72 = 576) and 720 lbs of dust ([18 - 8] x 72 = 720), uniformly mixed and distributed throughout its volume. The composition of the dust, shown in Table H-1, is modeled after that found at a representative steel mill.[3]

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[3] This composition is somewhat different than that listed in Appendix E-2, and is more representative of stainless steel rather than carbon steel melt shops. For the radionuclides of interest, however, the exact composition has a negligible effect on the external dose rates.

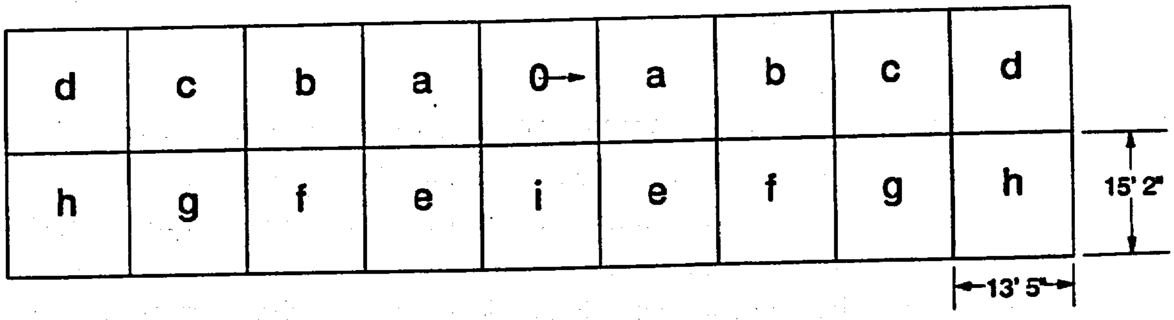


Figure H-6. Plan Drawing of Baghouse — Dimensions Are Typical of All Modules

Table H-1. Composition of Baghouse Dust

Compound	Percent Composition (by weight)
Fe_2O_3	54.5
CaO	24.7
Cr_2O_3	10.9
NiO	5.9
ZnO	3.0
PbO	1.0

The worker is assumed to be in the central module, marked "0" in the drawing, facing in the direction indicated by the arrow. The modules are separated by ¼-inch- thick steel walls—the other horizontal dimensions are shown in the drawing. The contribution of each module to the external exposure rate was calculated separately, using the dose conversion factors for anterior-posterior, posterior-anterior or lateral geometries, depending on whether the module is in front of, behind or alongside the worker. Module O was modeled as having the dust and the Nomex divided into two sources of equal size, with a 12-inch-wide space in the middle for the worker. The exposures from modules 0, a - d, and i were modeled assuming the worker was in the center of module 0. However, the contributions from modules e - h were calculated assuming the worker was at the wall separating module 0 from module i. The attenuation due to this wall was modeled assuming the radiation was normally incident on the wall, which results in less attenuation and therefore produces a somewhat more conservative result.

Exterior Maintenance. During the time the baghouse worker is performing outside maintenance and is monitoring the control panels, his external exposure would be from two sources: the half-full tank trailer that is normally parked under the baghouse, and the residual dust in the baghouse.

Exposure to Residual Dust in Baghouse. The bags are modeled as a rectangular solid source, 120 feet 9 inches long, 30 feet 4 inches wide, and 30 feet high, elevated 23 feet above ground level. In addition to the residual dust on the baghouse filters, an equal amount is assumed to have settled and collected on the floor of each module, which consists of a ⅝-inch-thick steel plate. This dust would thus form a layer 120 feet 9 inches long, 30 feet 4 inches wide and weighing 12,960 pounds (720 lb/module x 18 modules = 1,2960 lb). Since the worker moves around under the baghouse, his exposure was calculated along a line from the center to one corner, using Equations 6-5 and 6-6 in Chapter 6. The dose point is 1 m above ground.

Exposure to Tank Trailer. A tank trailer used to collect and transport baghouse dust is normally parked under one side of the baghouse. A description of the trailer was provided by David Fellows of the Mid West Region of The Heil Company, who also provided an engineering drawing which was the source of the illustration in Figure H-7. The trailer is approximately 29 feet long and 9½ feet in diameter. It was modeled as a semi-cylinder with a horizontal axis.

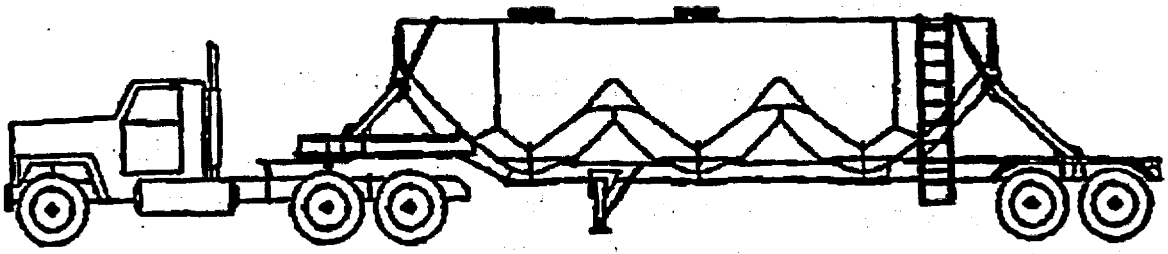


Figure H-7: The Heil Co., Super Jet Model 1040 Dry Bulk Trailer

Since the trailer arrives empty and leaves when it is full, it is modeled as being half-full on average. The mid-line of the load is 8 feet 8 inches above ground. The worker's position is assumed to vary uniformly over a range of 1 to 6 meters from the truck. The dust has an average bulk density of 57.5 lb/ft^3. The walls of the tank are aluminum, which would not significantly attenuate the penetrating γ-rays from the radionuclides in the dust for which external exposure is a significant pathway. The shielding due to the aluminum is therefore neglected.

Steel-Mill Duties Not Involving the Baghouse. Except on the days that he/she performs interior maintenance and during the one hour per day he/she spends on exterior maintenance, the baghouse worker performs other duties inside the mill. Since no particular mill worker is assigned to baghouse maintenance, the baghouse worker, during the time spent away from the baghouse, is assumed to have the same exposure rate as the crane operator, one of the three mill workers modeled.

H.6.2 Inhalation of Fugitive Emissions

While inside the baghouse, the worker is exposed to dust concentrations estimated to be 40 mg/m^3, with a respirable fraction of 0.76 (EPA 95). He/she wears a half-face mask, which has a rated filter efficiency of 90%. While monitoring the controls and performing maintenance outside the baghouse, he/she is exposed to an atmospheric dust loading of 1.2 mg/m^3, which is the reported dust concentration for a baghouse maintenance worker at an operating steel mill. While he/she performs duties away from the baghouse, the dust loading at his work station is assumed to be the average of the reported concentrations at nine other work stations at an operating steel mill.

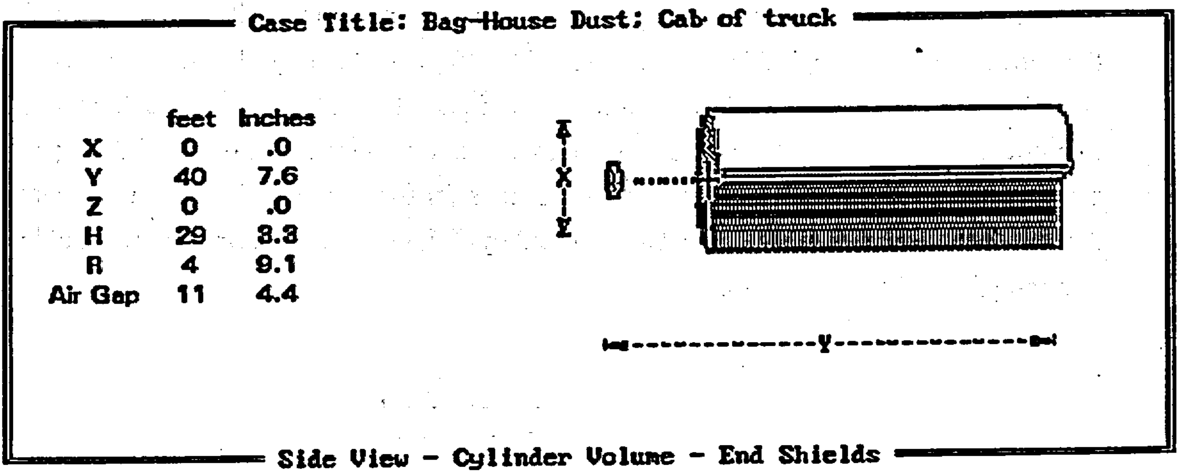
H.7 TRUCK DRIVER: BAGHOUSE DUST—DUSTDRIV

Since the truck driver transporting baghouse dust does not come in direct contact with the dust, his/her only significant exposure would be to direct penetrating radiation from the potentially contaminated dust inside the trailer.

H.7.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates to the baghouse dust truck driver. The load was modeled as described in Section H.6.1, above.

The position of the driver in the cab was scaled from the engineering drawing and determined to be 11 feet 4½ inches in front of the load. The model geometry is shown in Figure H-8.



Case Title: Bag-House Dust; Cab of truck

	feet	inches
X	0	.0
Y	40	7.6
Z	0	.0
H	29	8.3
R	4	9.1
Air Gap	11	4.4

Side View - Cylinder Volume - End Shields

Figure H-8: Baghouse Dust Truck Driver MicroShield Geometry

H.8 SLAG PILE WORKER—SLAGPILE

H.8.1 External Exposure

The external exposure to the slag pile worker was assessed using the FGR 12 dose coefficients, as discussed in Sections 6.3.1 and H.2.1, above. Since the worker is assumed to stand at the edge of the slag pile, his/her rate of exposure is one-half of what it would be in the center of the slag.

H.8.2 Inhalation of Slag Dust

The atmospheric dust concentration was estimated on the basis of actual field measurements performed as part of an EPA-sponsored study of fugitive emissions from slag loading operations (Bohn 78). In order to determine the emissions due to the loading operation, the investigators placed air samplers upwind from the emission source to determine the background concentration—i.e., dust concentrations in the air that are not attributable to the activity being monitored.

Six background dust concentration measurements were performed at a slag plant attached to a steel mill. The readings ranged from 0.5 to 3.2 mg/m^3, with an average of 2.6 mg/m^3. These measurements were made using a high-volume air sampler which is not sensitive to particles larger than about 30 µm. For the purpose of the exposure assessment, it is necessary to derive the concentration of respirable particles (AMAD < 10 µm). Although Bohn 78 does not present such data directly, the report shows that the ratio of particles with mass median diameters < 5 µm to particles < 30 µm varies from 0.27 to 0.31, with an average value of 0.29. EPA 95 presents a more detailed distribution of aerodynamic diameters for fugitive emissions from aggregate piles; these data were combined with the data reported by Bohn *et al.* to calculate the respirable fraction of slag dust as follows:

$$F_{10,B} = \frac{F_{10,E}}{F_{5,E}}\left(\frac{F_{5,B}}{F_{30,B}}\right)$$

$F_{10,B}$ = respirable fraction of fugitive dust, based on Bohn 78
= 0.51

$F_{10,E}$ = respirable fraction of fugitive dust, reported in EPA 95
= 0.35

$F_{5,E}$ = fraction of particles, AD < 5 µm, reported in EPA 95
= 0.20

$\frac{F_{5,B}}{F_{30,B}}$ = average ratio of F_5 to F_{30} reported in Bohn 78
= 0.29

H.9 SLAG USED IN ROAD CONSTRUCTION—SLAGROAD

The exposure time of the road construction worker depends on the fraction of slag generated by the melting of potentially contaminated scrap that is used in road construction during the peak year. This, in turn, depends on the rate of road construction and the production rate of slag at the reference steel mill. *Means Heavy Construction Cost Data* (Means 97), a standard reference for contractors, states that a road construction crew laying down a 300-mm (≈1-foot) deep pavement base of 40 mm crushed stone has a production rate of 1,505 m^2 per day. A crew laying down 100-mm (≈4-inch) thick asphaltic concrete has a rate of 3,462 m^2 per day.

Assuming that the same crew lays down both the pavement base and concrete, the area of road produced in a day can be determined as follows:

$$A = R_b x = R_c(1 - x) \qquad \text{(H-1)}$$

A = Rate of road production (m^2/d)

R_b = Production rate of road base
= 1505 m^2/d

R_c = Production rate of concrete pavement
= 3462 m^2/d

x = fraction of day spent laying down road base

Solving the two equations for x, we find

$$x = \frac{R_c}{R_b + R_c}$$

Substituting this expression in the first of Equations H-1, we obtain

$$A = \frac{R_b R_c}{R_b + R_c}$$

$$= 1049 \ m^2/d$$

The quantity of slag used per day can now be readily determined:

$$M = A\left(d_c f_c + d_b\right)\rho$$

M = rate of slag utilization
= 797.2 Mg/d
= 878.8 short tons/day

d_c = thickness of concrete
= 0.1 m

f_c = fraction of slag in asphaltic concrete
= 0.8

d_b = thickness of road base
= 0.3 m

ρ = bulk density of slag
= 2 g/cm^3

Since the reference steel mill has a melting capacity of 150,000 tons of steel per year, and since the mass fraction of slag, as listed in Section 6.2, is 0.117, the production rate of slag is 17,550 tons per year, or enough for about 20 days of road construction. Assuming an exposure duration of 7 hours per day, the road workers would be exposed for 140 hours per year.

H.9.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates for a worker using slag in road construction. This worker is assumed to be exposed to two primary sources: slag used in the road base and slag used as an aggregate in the concrete paving. The road was modeled as a rectangular solid source 4,000 meters (infinitely long), 36 feet wide and 6 inches thick, with a 1-foot-thick concrete cover.[4] The worker was assumed to be standing in the center of the road, the dose point being one meter above the surface. Advantage was taken of the symmetry of the source to make the best use of the computation time: instead of modeling the entire source volume with the dose point along the central axis, a source with one-half the width and one-half the height, with the dose point along one edge, was modeled. The calculated results were then multiplied by four to account for the three missing but identical quadrants. The model geometry of the road base is depicted in Figure H-9.

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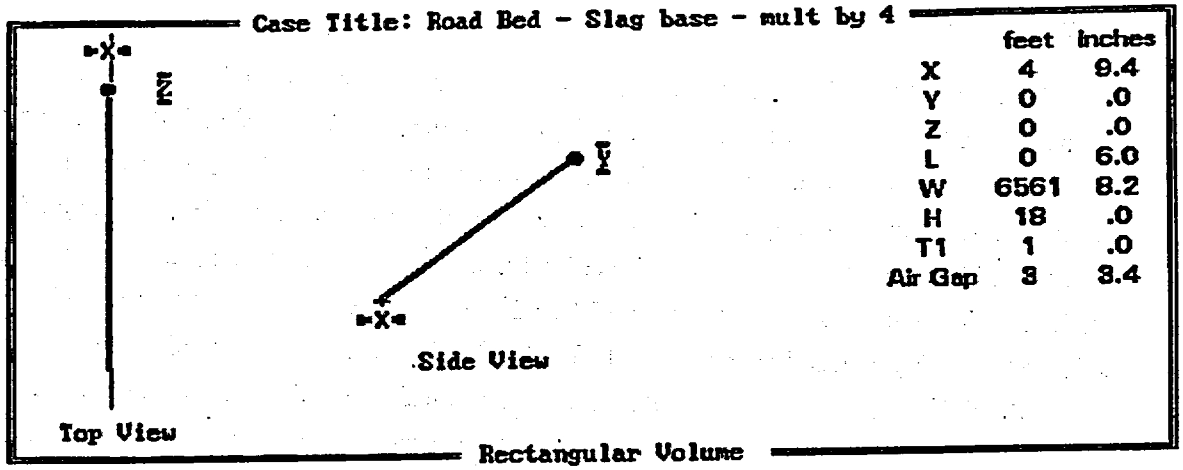
[4] These dimensions are taken from SC&A 93.



Case Title: Road Bed - Slag base - mult by 4

	feet	inches
X	4	9.4
Y	0	.0
Z	0	.0
L	0	6.0
W	6561	8.2
H	18	.0
T1	1	.0
Air Gap	3	3.4

Figure H-9: Slag Used in Road Base Construction MicroShield Geometry

Because of its thickness, density and area, the exposure rate from the concrete would not differ significantly from that of soil contaminated to an infinite depth. The external exposure from the concrete was therefore assessed using the FGR 12 dose coefficients, as discussed in Sections 6.3.1 and H.2.1, above. The calculated dose rates were multiplied by f_c, the fraction of slag in asphaltic concrete.

H.9.2 Inhalation of Slag Dust

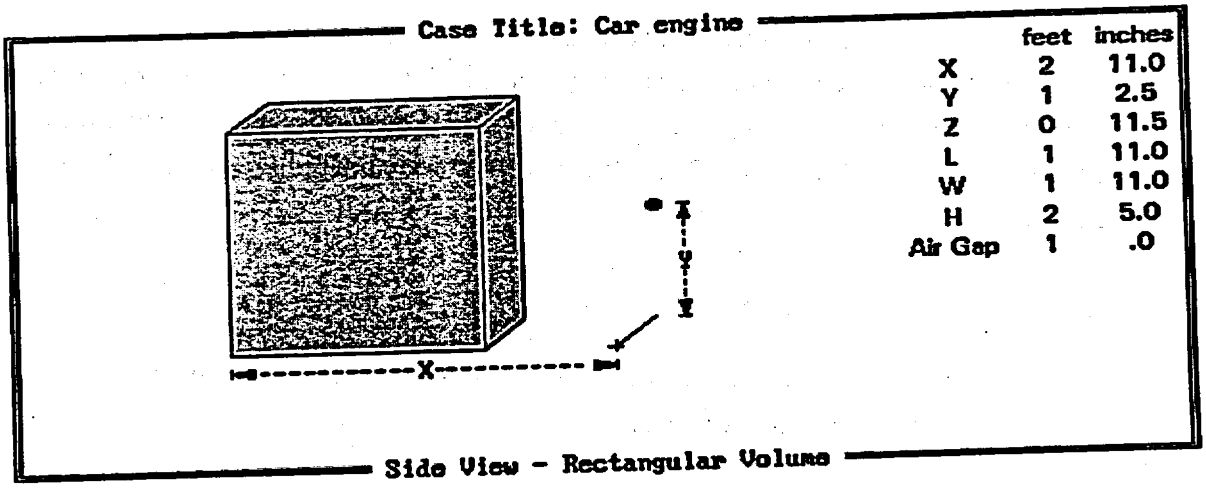
The road construction workers were assumed to be exposed to the same dust concentrations as the slag pile workers.

H.10 ASSEMBLING AUTOMOBILE ENGINES—ENGNWRKR

Because of his/her close proximity to a large mass of potentially contaminated metal, a worker assembling V-8 engine blocks was selected as the maximally exposed automobile worker. Since there is little opportunity for particulate matter to evolve from this operation, the only significant exposure pathway of this worker would be direct penetrating radiation from the cast iron block.

H.10.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates to an automobile engine assembler. The weight and dimensions of a typical V-8 engine were obtained from ADK, the engine rebuilder that formerly supplied rebuilt engines to Sears-Roebuck. The shipping weight of the engine is 350 pounds; the crate itself weighs about 5 pounds and has overall dimensions of 2 feet by 2 feet by 2½ feet. Assuming that the crate is one-half inch thick, the engine dimensions are 23 by 23 by 29 inches. The weight was divided by the volume to obtain an effective density of 0.632 g/cm^3. Since the worker would be moving back and forth while performing this task, dose rates were calculated at distances of 20 cm and 70 cm from the source. The average dose rates between these two distances were calculated using Equations 6-5 and 6-6 in Chapter 6. The model geometry for an intermediate distance is depicted in Figure H-10.



	feet	inches
X	2	11.0
Y	1	2.5
Z	0	11.5
L	1	11.0
W	1	11.0
H	2	5.0
Air Gap	1	.0

Figure H-10: Auto Engine Assembly MicroShield Geometry

H.11 MANUFACTURING INDUSTRIAL LATHES—LATHEMFG

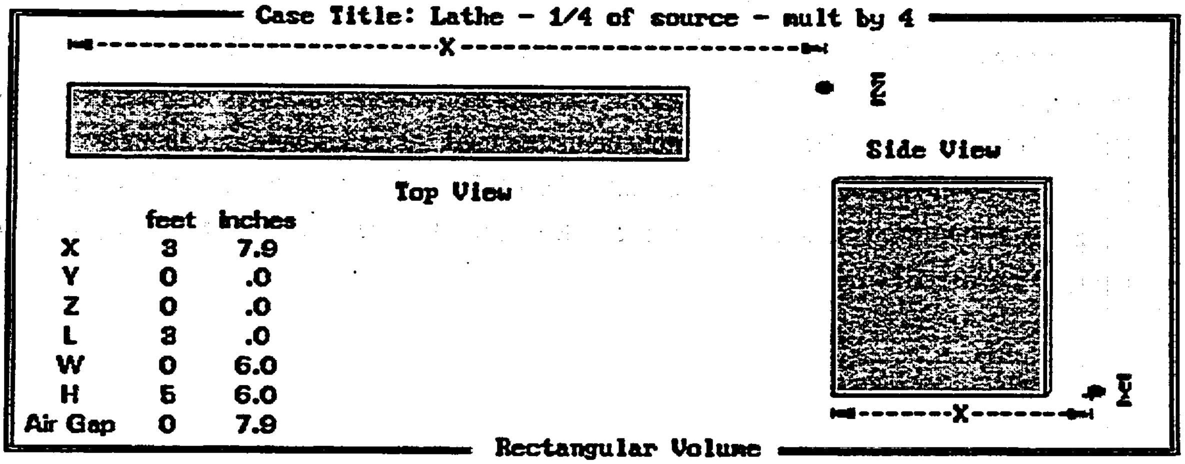
H.11.1 External Exposure

The MicroShield™ computer program was used to calculate normalized external dose rates to a worker manufacturing large industrial lathes. A large lathe observed in a commercial machine shop weighed 8 tons. The lathe bed, which would comprise most of this mass, was three feet wide and one foot thick. Assuming the bed contained all of the mass, it is calculated to be approximately 11 feet long. The lathe was thus modeled as a rectangular solid. Advantage was taken of the symmetry of the source to make the best use of the computation time: instead of modeling the entire source volume with the dose point along the central axis, a source with one-half the width and one-half the height, with the dose point along one edge, was modeled. The calculated results were then multiplied by four to account for the three missing but identical quadrants.

Since the worker would be moving back and forth while performing this task, dose rates were calculated at distances of 20 cm and 70 cm from the source. The average dose rates between these two distances were calculated using Equations 6-5 and 6-6 in Chapter 6. The model geometry for the 20-cm distance is depicted in Figure H-11.

H.11.2 Inhalation of Contaminated Dust

The grinding of the lathe bed could produce airborne dust. Newton *et al.* (1987) report that cutting metal with a side-arm grinder in a ventilated enclosure produced dust concentrations averaging 2.7 mg/m^3. This value was adopted for assessing the inhalation exposure of the lathe manufacturing worker.



Case Title: Lathe - 1/4 of source - mult by 4

	feet	inches
X	3	7.9
Y	0	.0
Z	0	.0
L	3	.0
W	0	6.0
H	5	6.0
Air Gap	0	7.9

Figure H-11: Lathe Manufacture MicroShield Geometry

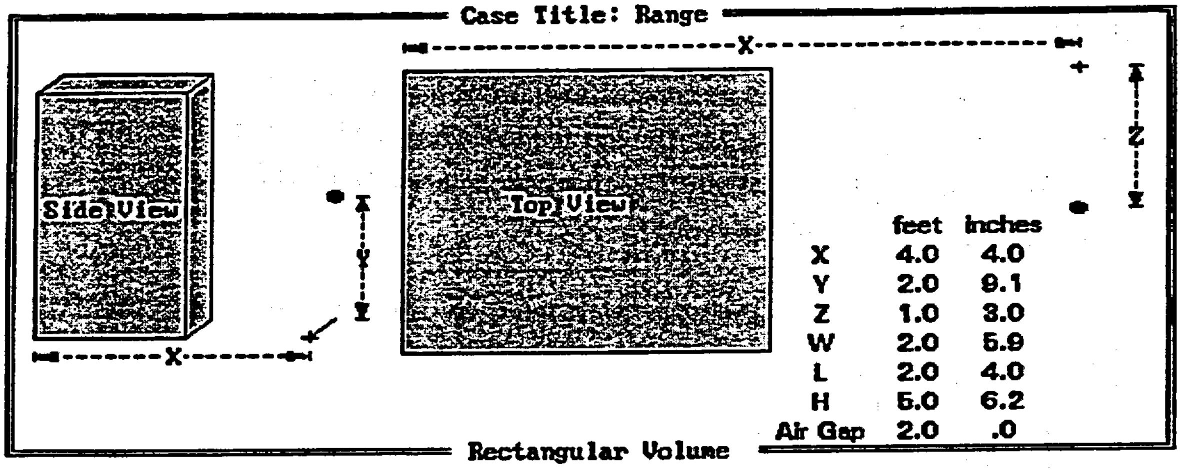
H.12 END-USER SCENARIOS

The scenarios describing the exposures of the end users of finished products have several features in common. First, the maximally exposed user of a given product is assumed to use a product made entirely of potentially contaminated scrap metal. While it is implausible that a lathe fabricator would be exposed during an entire year to cast iron that was made entirely of potentially contaminated scrap metal, for instance, it is reasonable to believe that at least one lathe made from such metal could be produced. Since the lathe operator could be assigned to the same machine for one year, he/she would be exposed to such a source time. The same is true for the other products, all of which have useful lives of more than one year.

The second distinguishing feature of the end-user scenarios is that, since the user would have the same product for at least a year, the radionuclides would be decaying during this time. Consequently, Equation 6-9 in Section 6.3.4, which explicitly accounts for radioactive decay, is used to calculate the dose during that year. Finally, since no significant erosion of the metal in the finished product is expected in normal use, there are no significant internal exposure pathways, except for the potential contaminants leached from the cast iron frying pan.

H.12.1 Kitchen Range User—COOKRNGE

The MicroShield™ computer program was used to calculate normalized external dose rates to a user of a large kitchen range, modeled after a Sears Kenmore 30-inch double oven, model No. 78509. Its overall size is approximately 66 inches high, 29 inches wide and 28 inches deep; it weighs 284 pounds. The effective density of 0.1417 g/cm^3 was calculated by dividing the weight by the volume. The dose point is two feet in front of the source. The model geometry is depicted in Figure H-12.



	feet	inches
X	4.0	4.0
Y	2.0	9.1
Z	1.0	3.0
W	2.0	5.9
L	2.0	4.0
H	5.0	6.2
Air Gap	2.0	.0

Figure H-12: Range User MicroShield Geometry

H.12.2 Taxi Driver—TAXIDRVR

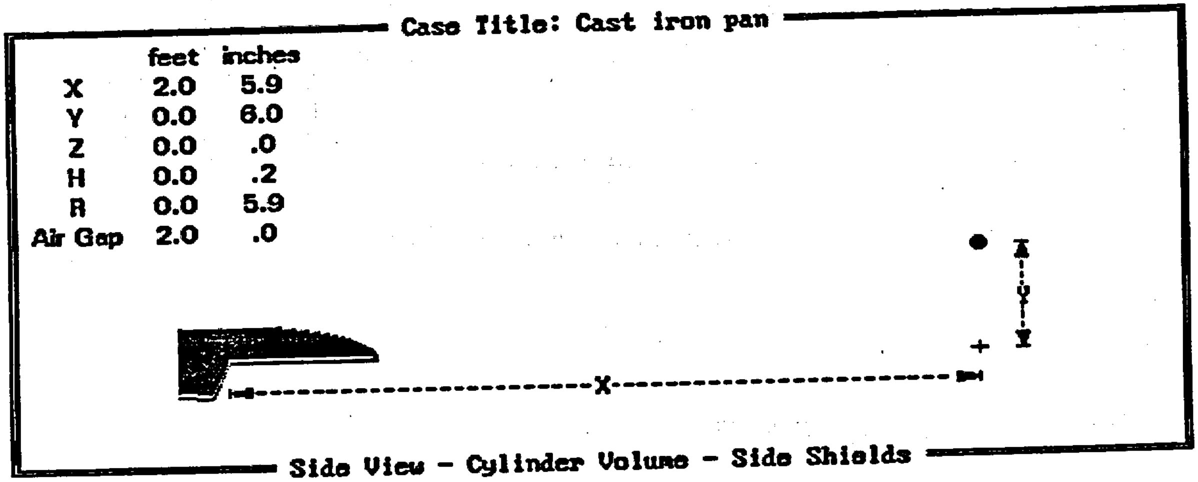
The maximally exposed taxi driver is assumed to be an owner/operator who drives a taxi with a body shell made of sheet metal that was made entirely of potentially contaminated scrap metal. The MicroShield™ computer program was used to calculate normalized external dose rates to this driver. The dimensions were based on a Ford Taurus, a widely sold mid-sized American-made automobile. Based on data and a photograph published in the April, 1996 issue of *Consumer Reports*, the interior of the car was modeled as a steel box, 6 feet wide, 4 feet high and 9 feet from front to back. The total weight is estimated to be 900 lbs. This assessment is somewhat conservative, since the boundary of the passenger compartment is assumed to comprise the entire mass of the shell. In reality, a this mass also comprises the outer skin of the engine and trunk compartments, which are further from the driver and would thus make a smaller contribution to the dose.

H.12.3 Lathe Operator—OP-LATHE

The normalized external dose rates to the operator of a large industrial lathe are calculated using the same geometry as that described in Section H-11 for the lathe manufacturing worker.

H.12.4 Cooking on a Cast Iron Pan—FEFRYPAN

The MicroShield™ program was used to calculate normalized external dose rate to a person cooking with a cast iron frying pan. The pan was modeled as a flat disc about 12 inches in diameter and weighing about six pounds. The dose point is two feet from the edge of the pan. The model geometry is depicted in Figure H-13.


Case Title: Cast iron pan

	feet	inches
X	2.0	5.9
Y	0.0	6.0
Z	0.0	.0
H	0.0	.2
R	0.0	5.9
Air Gap	2.0	.0

Side View - Cylinder Volume - Side Shields

Figure H-13: Frying Pan User MicroShield Geometry

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APPENDIX H-1

NEARBY RESIDENT EXPOSED TO EFFLUENT AIRBORNE EMISSIONS

SYNOPSES OF CAP-88 ANALYSES

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                    C A P 8 8 - P C

                     Version 2.00

          Clean Air Act Assessment Package - 1988


                 S Y N O P S I S   R E P O R T

               Non-Radon Individual Assessment
                 Oct  27, 1996  04:09 am


Facility:  C-14 Recycle - HAR0631
 Address:
    City:
   State:  IL          Zip:


Source Category:
    Source Type:  Stack
  Emission Year:  1996


Comments:  RSM Recycle Doses from Airborne C-14 Release
           Check of GENII results


                Effective Dose Equivalent
                      (mrem/year)
               ______________________________

                        8.66E-04
               ______________________________

At This Location:    1000 Meters East

    Dataset Name:   RSM C-14 HAR0631
    Dataset Date:   Oct  27, 1996  04:07 am
       Wind File:   C:\CAP88PC2\WNDFILES\HAR0631.WND
```

MAXIMALLY EXPOSED INDIVIDUAL

Location Of The Individual:	1000 Meters East
Lifetime Fatal Cancer Risk:	2.11E-08

ORGAN DOSE EQUIVALENT SUMMARY

Organ	Dose Equivalent (mrem/y)
GONADS	3.72E-04
BREAST	9.70E-04
R MAR	1.71E-03
LUNGS	4.52E-04
THYROID	4.49E-04
ENDOST	3.56E-03
RMNDR	8.26E-04
EFFEC	8.66E-04

```
                    RADIONUCLIDE EMISSIONS DURING THE YEAR 1996

                         Source
                           #1     TOTAL
Nuclide  Class  Size     Ci/y     Ci/y
_______  _____  ____   _______  _______

C-14       *    0.00   1.1E-02  1.1E-02


                    SITE INFORMATION

                         Temperature:    10 degrees C
                       Precipitation:   100 cm/y
                       Mixing Height:  1000 m
```

```
                     SOURCE INFORMATION

     Source Number:      1
                       -----

   Stack Height (m):       0.
       Diameter (m):       0.

    Plume Rise
  Pasquill Cat:     A        B        C        D        E        F        G
                 -------  -------  -------  -------  -------  -------  -------

         Zero:        0.       0.       0.       0.       0.       0.       0.


                     AGRICULTURAL DATA
                                              Vegetable    Milk     Meat
                                              ---------   -----    -----

                     Fraction Home Produced:    0.700      0.399    0.442
             Fraction From Assessment Area:     0.300      0.601    0.558
                         Fraction Imported:     0.000      0.000    0.000

                     Food Arrays were not generated for this run.
                                Default Values used.


  DISTANCES (M) USED FOR MAXIMUM INDIVIDUAL ASSESSMENT

     1000
```

```
                         C A P 8 8 - P C

                          Version 2.00

             Clean Air Act Assessment Package - 1988


                   S Y N O P S I S   R E P O R T

                 Non-Radon Individual Assessment
                     Oct  27, 1996  04:11 am


     Facility:  RSM Recycle - LAX0304
      Address:
         City:
        State:  IL          Zip:


     Source Category:
         Source Type:  Stack
       Emission Year:  1996


     Comments:  RSM Recycle Doses from Airborne I-129 Release
                Check of GENII results


                 Effective Dose Equivalent
                        (mrem/year)

                 _________________________

                         7.91E-01
                 _________________________


    At This Location:   1000 Meters East

        Dataset Name:  RSM I-129 LAX
        Dataset Date:  Oct  27, 1996  03:57 am
           Wind File:  C:\CAP88PC2\WNDFILES\LAX0304.WND
```

MAXIMALLY EXPOSED INDIVIDUAL

Location Of The Individual:	1000 Meters East
Lifetime Fatal Cancer Risk:	4.65E-06

ORGAN DOSE EQUIVALENT SUMMARY

Organ	Dose Equivalent (mrem/y)
GONADS	6.68E-02
BREAST	1.04E-01
R MAR	1.08E-02
LUNGS	2.42E-02
THYROID	2.48E+01
ENDOST	4.14E-02
RMNDR	2.75E-02
EFFEC	7.91E-01

```
                 RADIONUCLIDE EMISSIONS DURING THE YEAR 1996

                          Source
                            #1    TOTAL
Nuclide Class Size        Ci/y     Ci/y
_______ _____ ____      _______  _______

I-129     D   1.00      1.5E-02  1.5E-02


                          SITE INFORMATION

                              Temperature:     10 degrees C
                            Precipitation:    100 cm/y
                           Mixing Height:   1000 m
```

```
                                                           SYNOPSIS
Oct  27, 1996  04:11 am                                    Page  3

                      SOURCE INFORMATION

          Source Number:   1
                         ____

     Stack Height (m):      0.
         Diameter (m):      0.

       Plume Rise
    Pasquill Cat:    A      B      C      D      E      F      G
                   _____  _____  _____  _____  _____  _____  _____

            Zero:    0.     0.     0.     0.     0.     0.     0.


                      AGRICULTURAL DATA

                                         Vegetable    Milk     Meat
                                         _________    ____     ____

               Fraction Home Produced:     0.700      0.399    0.442
          Fraction From Assessment Area:   0.300      0.601    0.558
                   Fraction Imported:      0.000      0.000    0.000

                      Food Arrays were not generated for this run.
                               Default Values used.

     DISTANCES (M) USED FOR MAXIMUM INDIVIDUAL ASSESSMENT

        1000
```

APPENDIX H-2

EXPOSURE FROM THE USE OF SLAG IN AGRICULTURE

March 7, 1997

To: W-A Rad 5-07 File

From: John Mauro

Subject: The Slag Agricultural Pathway

During the review of the draft TSD, EPA inquired whether we should have included in the RME individual dose assessment the dose from using slag as an agricultural conditioner. The following presents an assessment of the potential significance of this pathway.

Because of its high lime content (up to 50%), slag can be used as a soil conditioner. In general, 50 to 100 lbs of lime is applied to 1000 ft² of soil for pH adjustment. Assuming a plow depth of 15 cm and soil density of 1.6 g/cm³, the normalized dose to the RMEI via this pathway can be approximated as follows:

$$D_{ia} = c_{ig} D_{is} f_c f_{gs}$$

D_{ia} = normalized dose from radionuclide *i* via the slag agricultural pathway (mrem/y per pCi/g in scrap)

c_{ig} = concentration factor of radionuclide *i* in slag (see Table 6-3)

D_{is} = normalized dose from radionuclide *i* via the soil agricultural pathway (mrem/y per pCi/g in soil—EPA 94, Table 3-1)

f_c = fraction of slag from potentially contaminated scrap
= 0.11

f_{gs} = fraction of slag in soil (by weight)

$$= \frac{m_g}{A\,\rho_s\,d_s}$$

m_g = mass of slag
= 100 lb
= 4.54×10^4 g
A = 1000 ft²
= 9.29×10^5 cm²
ρ_s = soil density
= 1.6 g/cm³
d_s = plow depth of soil layer
= 15 cm

The results for radionuclides that concentrate in the slag are presented in the following table. The column headings correspond to the terms defined above.

Table H2-1.
Comparison of Normalized Annual Doses via Agricultural Slag Pathway with Doses to RMEI

Radionuclide	D_{ia}	D_{is}	RMEI dose[a] (mrem/y per pCi/g)
Nb-94	Neg	Neg	1.33
Ce-144	1.2E-2	2.09e-05	4.6E-2
Eu-152	Neg	Neg	9.61E-1
Ra-226	4.35	7.58e-03	1.61
Ra-228	1.6	2.79e-03	0.895
Th-228	7E-2	1.22e-04	2.24
Pm-147	3E-4	5.23e-07	1.46E-4
Th-229	.014	2.44e-05	4.51
Th-230	1.5	2.61e-03	0.642
Th-232	2.1	3.66e-03	2.84
Pa-231	5.1	8.89e-03	2.53
U-234	0.16	2.79e-04	0.314
U-235	0.12	2.09e-04	0.395
U-238	0.16	2.79e-04	0.302
Np-237	9.6	1.67e-02	1.62
Pu-239	0.7	1.22e-03	0.729
Am-241	0.08	1.39e-04	1.22
Cm-244	0.21	3.66e-04	0.675
Sr-90	5	8.72e-03	3.03

[a] Table 7-1

These results show that the reasonable maximum dose via the agricultural slag pathway is a small fraction of the dose to the RMEI for each of the nuclides listed.

APPENDIX I

LEACHING OF RADIONUCLIDES FROM SLAGS

LEACHING OF RADIONUCLIDES FROM SLAGS

Steelmaking slags are typically composed of calcium silicates and aluminoferrites together with fused oxides of calcium, iron, manganese, and magnesium (NSA94). Based on a 1991 survey of member companies the National Slag Association quoted the average chemistry for steel slags as:

CaO - 42.88%
SiO_2 - 14.89%
MgO - 8.14%
MnO - 5%
FeO - 25%
P_2O_5 - 0.8%
S - 0.078%
Al_2O_3 - 5.00%
Moisture - 3.60%

As described in Appendix E, a number of radionuclides are expected to partition strongly to the slag during the EAF melting of contaminated carbon steel. Typically, this slag is stored for at the steel mill for a period of up to several months before disposal. Ultimate disposal generally involves use in road fill and as an aggregate in building products. In 1992, 6.9 million metric tons of steel slag were sold or used in the U.S. for the following purposes (SOL93):

- Asphaltic concrete aggregate - 13%
- Fill - 16%
- Road base - 35%
- Railroad ballast - 3%
- Soil conditioning, ice control, misc. - 33%

According to the U.S. Geological Survey[1], there are currently 13 firms which process steel slags at 76 facilities in 28 states (USG96). In 1995, 85% of all iron and steel slags were shipped by truck with an average shipment range of 30 miles; 4% were shipped by water with an average range of 250 miles; and 4% by rail with an average range of 175 miles. The balance of the slag (7%) was used at the plant sites.

---

[1] This data collection and analysis function was handled by the Bureau of Mines prior to 1996.

During storage and use (or disposal), the slag will be subjected to weathering and certain components may be leached from the slag and ultimately contaminate the local groundwater. This Appendix presents the limited information uncovered in this study which can be used to model the leaching of radionuclides which partition to the slag.

I.1 SLAG CEMENT LEACHING STUDIES

The American Nuclear Society has developed and formalized detailed procedures for measuring the leachability of solidified low-level radioactive wastes (ANS86). This procedure involves testing of controlled geometry specimens in demineralized water at 17.5 to 27.5°C to determine the release during individual time steps and cumulatively. Mass transport is assumed to be controlled by a diffusion process. When the fraction leached from a uniform sample is less than 20%, behavior can be approximated by a semi-infinite medium where the "effective diffusivity" is given by the following equation:

$$D = \pi T \left(\frac{a_n V}{A_o \Delta_{nt} S} \right)^2 \tag{1}$$

where:

D = effective diffusivity, cm^2/s

V = specimen volume, cm^3

S = geometric surface area, cm^2

A_o = total activity of a given nuclide at $t = 0$

a_n = activity of nuclide released during time interval n

$\Delta_n t$ = $t_n - t_{n-1}$, duration of nth leaching interval,

T = mean time of the leaching interval

$= \left(\frac{\sqrt{t} + \sqrt{t_{n-1}}}{2} \right)^2$

When the cumulative fraction leached, $\sum_n \frac{a_n}{A_o}$, is greater than 20%, corrections must be made to Equation 1 for specimen geometry.

Using a model and procedures similar to those described in ANS86, Japanese investigators have determined the fractional leaching of Sr-90, Co-60, Cs-137, and H-3 from cement/slag composites (MAT77, MAT77a, MAT79) in deionized water and synthetic sea water. The duration of the leaching tests was about 100 days. The radionuclides were incorporated into the cement via a sodium sulfate solution. The composition of the slag cement (wt %) was as follows:

SiO_2 - 28.7
Al_2O_3 - 11.5
Fe_2O_3 - 2.3
CaO - 50.9
MgO - 3.2
Insoluble Residue - 0.8
Ignition Loss - 0.6

Leaching data were analyzed using a plane source diffusion model to derive the expression

$$f=2S\sqrt{Dt}/V\sqrt{\pi} \qquad (2)$$

where f is the fraction of the radionuclide leached in t days, S and V are the specimen surface area and volume, respectively, and D is the diffusion coefficient in cm^2/day. The diffusion coefficient is obtained from the slope m of the linear relation between f and $\sqrt{t}$ as follows:

$$D = \pi m^2 V^2/4S^2 \qquad (3)$$

Since the actual leaching process involves an initial rapid leaching rate of a few days (ca. seven days for Sr-90 and two days for Co-60) duration followed by a longer term linear relation between f and $\sqrt{t}$, the experimental data are fitted to an equation of the form

$$f=m\sqrt{t}+\alpha \qquad (4)$$

Because of certain limitations and problems such as the initial leach rate, Matsuzuru et al. defined L, the leaching coefficient, with the same mathematical form as D in equation 3.

Adjustments to the fraction leached for various geometries can made using the following expression:

$$f_x = f_y \cdot (S/V)_x/(S/V)_y \qquad (5)$$

I.1.1 Strontium-90 (MAT77a)

Values of L (cm^2/day) for Sr-90 leaching from slag cements ranged from 1.2 to 1.7 x 10^{-7} for both deionized water and synthetic sea water at 25°C. Using average values of L for samples cured 7 days prior to test in deionized water, a surface area of 94 cm^2 and a volume of 70 cm^3, SC&A developed the following equation for the fractional leaching:

$$f=5.8\times 10^{-4}\sqrt{t}+4.97\times 10^{-3} \qquad (6)$$

From equation 6, f would be 1.6% after 365 days.

The leachability of the Sr-90 was reported to be about 1/10th that of Cs-137.

I.1.2 Cobalt-60 (MAT77)

Values of L (cm^2/day) for Co-60 leaching from slag cements ranged from 9.83 x 10^{-10} to 1.89 x 10^{-9} for both deionized and synthetic sea water at 25°C. Using the same principles as for Sr-90 above, the fractional leaching is

$$f = 4.9 \times 10^{-5}\sqrt{t} + 4.33 \times 10^{-4} \qquad (7)$$

The amount of Co-60 leaching based on equation 7 would be about 0.14% in 365 days.

Matsuzuru et al. observed that the quantity of Co-60 leached during the initial 2-day period of accelerated leaching was comparable to that leached over the next 98 days where the $\sqrt{t}$ dependency was observed.

The leaching coefficient of Co-60 was found to be 10^3 to 10^5 lower than for Cs-137.

I.1.3 Tritium (MAT79)

In their tritium studies, Matsuzuru, et al. considered the initial period of accelerated leaching more rigorously than in previous studies defining the initial rate by the equation

$$f_i = m_i\sqrt{t} \tag{8}$$

where the subscript i refers to the initial leach rate. Subsequent leaching was described by equation 4 above. Leaching coefficients (based on equation 3) in sea water and deionized water at 25°C for samples with seven-days curing ranged from 1.06×10^{-4} to 2.05×10^{-4} cm^2/day. The fractional release equation is

$$f = 0.018\sqrt{t} + 0.156 \tag{9}$$

and the release from a sample 4.5 cm in diameter by 4.4 cm high is about 50% in one year.

I.2 SLAG LEACHING STUDIES

This section describes leaching studies done on pure slags rather than slag/cement composites.

Australian researchers at CSIRO incorporated the toxic elements As, Sb, Cd, Zn, and Cr into various types of slags by melting at 1300°C and subsequently leached the slags according to the EPA TCLP protocol (JAH94). In the TCLP test, a sample of at least 100 g, which has a minimum surface area of 3.1 cm^2/g or passes through a 9.5 mm sieve, is treated with about 2,000 g of extractant for 18+/-2 hours at 22+/-3°C using rotary agitation. The extractant has a pH of either 4.93 or 2.88 depending on the basicity of the sample (40 CFR 261, Appendix II, Method 1311). The pH is achieved by use of acetic acid which is buffered with sodium acetate for the higher pH level (55 FR 11798).

Slag samples were prepared by both slow cooling and quenching. Examination of the slag samples with an optical microscope showed that interconnecting porosity was present in the slow cooled and most of the quenched samples. Slow-cooled slag samples were crushed to either a "coarse" size (100% minus 10 mm) or a "fine" size (100% minus 1 mm) for the leaching tests. In generalizing on the results of the TCLP tests, the researchers observed that

- As and Sb leached more readily than Cd, Cr, and Zn
- Fine particles generally leached more readily than coarse particles
- Slow cooled samples showed similar behavior to quenched samples

Based on the information presented in JAH94, SC&A estimated the fraction leached using the following assumptions:

- Slag compositions from Table III of JAH94
- Sample size - 100 g
- Extractant volume - 2 L

Results are presented in Table I-1. For three of the slags (CaFe1, CaFeSi1, and FeSi1), the compositions are markedly dissimilar to those expected from EAF melting of carbon steel. The other three slags, while not identical to EAF slags, are useful for developing preliminary modeling parameters. Unfortunately, of the five elements studied, only Cr is expected to be partition to the slag found in any significant quantity. However, in the absence of element specific leaching data, Cr can be considered as a surrogate for the stable oxides expected in slags. Assuming that the fraction leached is proportional to $\sqrt{t}$, the fraction leached can be expressed by the equation

$$f = m\sqrt{t} \qquad (10)$$

where the upper limit of m is about $7 \times 10^{-6}/(\text{day})^{0.5}$ (based on Cr in the BF2 slags and an 18-hr leach test).

Table I-1. Fraction of Various Toxic Elements Leached from Slags Using EPA TCLP Protocol

Slag	Fraction Leached				
	As	Sb	Cd	Cr	Zn
CaFe1	3.48E-03	4.21E-05	3.10E-04	0.00E+00	3.00E-05
CaFeSi1	3.53E-03	2.68E-04	2.40E-04	0.00E+00	2.70E-05
CaFeSi2	5.09E-04	2.37E-04	6.80E-05	5.63E-07	2.3E-05
FeSi1	1.54E-04	1.10E-04	1.15E-04	4.82E-07	2.30E-5
BF1	1.68E-04	1.03E-04	1.10E-04	0.00E+00	1.34E-04
BF2	9.80E-04	4.29E-04	1.20E-03	6.00E-06	1.23E-03

The U.S. Army Corps of Engineers has extensively used slags for fill and bank erosion protection in the upper Ohio River Valley drainage basin. Because of concerns about what elements might leach from the slags, the Corps of Engineers conducted a series of slag leaching experiments (USA89). Two types of experiments were conducted; one involving experimental weathering beds and the other involving laboratory elutions. In the weathering bed experiments, slag samples weighing 40 to 75 lbs were placed in Nalgene containers and exposed to atmospheric weathering for 980 days. The leachate (i.e., rain water and snow) passing through the slag beds was collected and periodically analyzed (eleven different times) to determine the quantities of various elements leached from the slag. For the laboratory elution experiments, 2.2-lb samples of weathered slag were collected at the same times as the leachate samples and mixed with distilled water. These laboratory samples were then eluted for 109 to 198 hours with periodic stirring. The elutriate was analyzed for the same species as the leachate from the weathering tests. Elution tests were also conducted on unweathered samples.
Five types of slag were tested including:

- Three air-cooled blast furnace slags (Slags 1A, 1B, and 1C)
- One mixed slag - ca. 50% BOF and 50% EAF (Slag 2)
- One slag mixture - BOF, EAF, blast furnace, foundry waste and fire brick (Slag 3)

The discussion which follows focuses on the mixed BOF/EAF slag (slag 2) since it is deemed to be most relevant to expected leaching behavior of EAF slags. Slag 2 was in the form of gravel with 99.9% being between 2.38 and 4.76 mm and was two days old at the time of collection. Measurements are summarized in Table I-2.

Table I-2. Constituents Leached from Slag 2 (USA898)

Parameter	Unweathered Slag Elutriate	Weathered Slag Elutriate	Weathering Bed Leachate		
	(mean)	(mean)	(initial)	(mean)	(final)
Total P (µg/l)	17	12	L10[1]	L10	L10
Total Ca (mg/l)	108	63	22	20	27
Total Mg (mg/l)	8	3	13	15	12
Total Na (mg/l)	2	1	44	8	1
Total K (mg/l)	2	2	14	4	L1
Total Ba (µg/l)	35	23	L10	68	L10
Total Be (µg/l)	12	1	L1	L1	L1
Total Cd (µg/l)	1	1	L1	2.3	L1
Total Cr (µg/l)	130	100	38	24	21
Total Cu (µg/l)	17	12	L5	10	8
Total Fe (µg/l)	18,286	4,038	L100	L100	L100
Total Mn (µg/l)	1,464	352	L10	L10	L10
Total Ni (µg/l)	114	40	39	9	L5
Total Pb (µg/l)	11	16	L2	2	L2
Total Zn (µg/l)	60	L50	L50	66	L50
Total Sb (µg/l)	L100	L100	L100	L100	L100
Total Al (µg/l)	7868	950	L50	99	70
pH	9.8	9.6	8.4	8.2	8.6

1 - "L" means less than

It can be seen from Table I-2 that, in a number of instances, the elutriate from the weathered slag contains significantly higher contaminant levels than does the weathered slag leachate. The Corps of Engineers observed that:

> Standard and modified slag elutriate procedures provide some insights into worst case scenarios that might occur during and immediately following placement of disturbed slags and can provide some general ideas about slag reactivity and leachate composition. These procedures, however, can very grossly exaggerate the potential of stabilized slags to leach metals and otherwise have serious limitations in providing a basis for predicting long term leachate quality.

The leaching process was temperature dependent with higher concentrations of contaminants detected in samples taken during the summer months. However, over the 980-day duration of the tests, the concentration appeared to be independent of time with the exception of K, Na, and Ni. The temporal concentration dependence for Ba, Ca, Cr, K, and Mn is shown in Figures I-1 and I-2. Temporal variations for other sampled elements were not included in USA89.

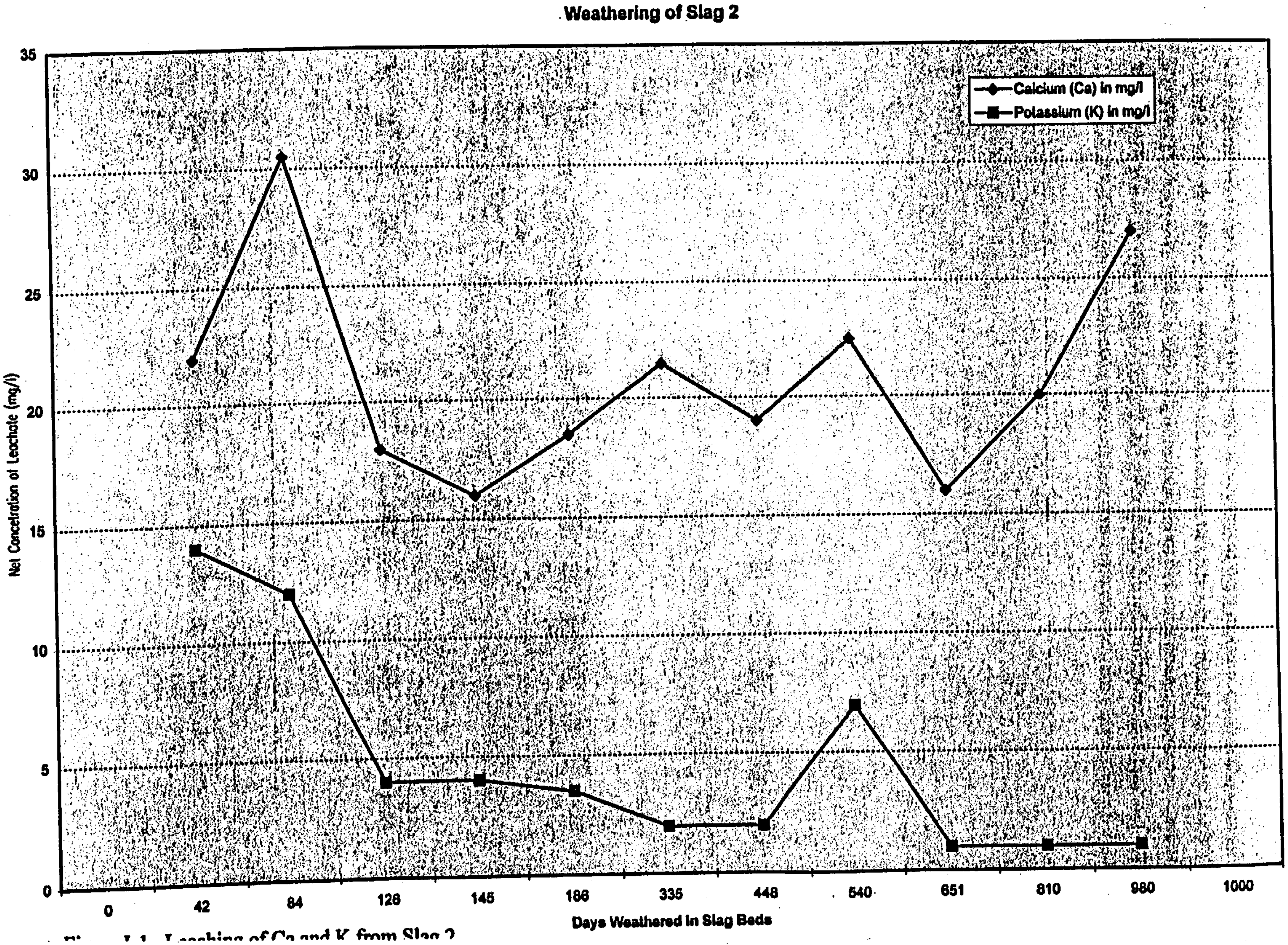

Figure L-1. Leaching of Ca and K from Slag 2

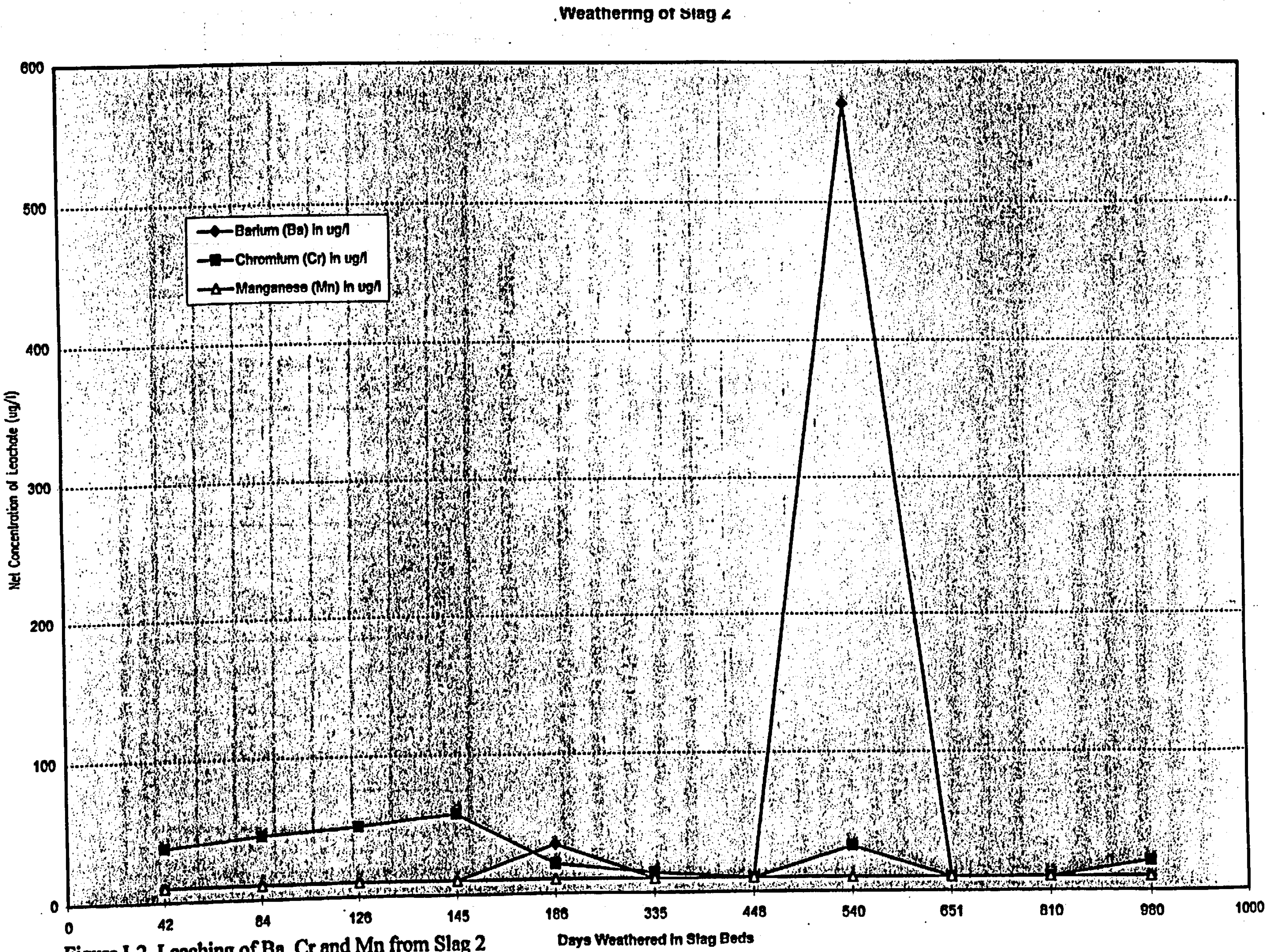

Figure I-2. Leaching of Ba, Cr and Mn from Slag 2

Emery of McMaster University also examined the leaching of toxic elements from slags (EME80). He noted that "Leachates from steel slags do not contain significant concentration of toxic constituents, but, in stagnant water conditions, deposits of calcite have been noted." He also observed that slags could undergo a potential volume expansion of up to 10% due to hydration of free calcium and magnesium oxides.

Emery quoted solubility data based on an early EPA procedure of mixing two parts distilled water and one part slag and gently agitating for 72 hours. The following leachate concentrations were cited for an electric arc furnace slag:

Cr - 0.27 mg/l
Cu - <0.03 mg/l
F - 1.5 mg/l
Mn - <0.01 mg/l
Pb - 0.44 mg/l
Zn - <0.01 mg/l
pH - 12.4

Emery also obtained data on filtrates from blast furnace slag sampled every 24 hours for five days. Results are presented in Table I-3. The slags were crushed to -13mm/+300 μm, and vigorously agitated at 3 Hz (60 g slag/3 liters of water). Tests were run on freshly-produced slag (F) and slag (A) which had been stored in a small pile for two years. Unlike the EAF slag results noted above, the pH of the blast furnace leachates varied from 7.6 to 7.9.

Table I-3. Blast Furnace Slag Solubility Data (EME80)

Slag	Analysis of Filtrate (mg/l)[a]					
	Ca	Cu	Fe	Mg	Ni	Pb
F-1[b]	13.5	0.02	0.02	0.60	<0.01	<0.01
F-3	7.2	<0.01	0.04	0.84	<0.01	0.02
F-5	6.6	<0.01	0.02	1.08	0.03	<0.01
A-1	15.9	0.05	0.03	0.60	<0.01	<0.01
A-3	7.5	<0.01	0.07	0.76	<0.01	<0.01
A-5	8.0	0.04	0.05	0.87	<0.01	<0.01

a - Cr, Mn, and Zn less than 0.01 mg/l in all cases
b - F-1 sample from first 24-hour period, F-3 from third 24-hour period, etc.

With the exception of Fe and Mn, Emery's results agree within an order of magnitude of those of the Corps of Engineers (see unweathered slag elutriate in Table I-2).

According to West of International Mill Services (a major slag dealer), all slags which they handle meet the TCLP test limits by at least an order of magnitude (WES96). Regulatory levels for the test (in mg/l) are: As - 5, Ba - 100, Cd - 1, Cr - 5, Pb - 5, Hg - 0.2, Se - 1, Ag - 5.

Pillai and Pandey considered the use of slag for the removal of undesirable ions in water treatment plants (PIL89). In support of this activity, they determined the extent to which minor elements were leached from the slag. Chemical analysis of minor elements in slag leachates determined after holding five-gram slag samples in 50 ml distilled water overnight and sampling the filtrate. Cu, Co, Ni, Pb, Zn, Bi, Cd, Cr, Sb, Be, Mo, V, Li, and Rb were found in both blast furnace and open hearth furnace slags at ppm levels, but <u>none</u> found were in slag leachates. The water soluble fraction of open-hearth and blast-furnace slags was 0.83 and 0.80%, respectively. The water soluble components are mainly alkali and alkaline earth metals.

In its 1990 report to Congress on mineral processing wastes, EPA described leachate analyses obtained from BOF slags leached by the EP or SPLP tests (EPA90)[2]. Results are presented in Table I-4.

---

[2] This study considered only blast furnace and BOF wastes; EAF wastes were not addressed.

Table I-4. Constituents of Concern in Steel Furnace Slag Leachates

Potential Constituents of Concern	No. of Times Constituent Detected/No. of Analyses for Constituent[a]	Screening Criteria (µg/l)	No. of Analyses Exceeding Criteria/No. of Analyses for Constituent
Manganese	3/6	500	3/6
Fluoride	1/1	21,000 40,000	1/1 1/1
Arsenic	3/8	2[b] 500	3/8 1/8
Lead	4/14	210 50 320	3/14 4/14 3/14
Silver	2/14	12	2/14
Iron	3/6	3,000	1/6
Molybdenum	2/8	100	1/8
Barium	7/14	18,000 10,000	1/14 1/14

a - Based on EP leach test except As which is based on SPLP test.
b - Based on 1 x 10^{-5} lifetime cancer risk.

EPA made the following observations about exposure potential from slags (EPA90).

> In theory, constituents of potential concern in blast furnace and steel furnace slag could enter surface waters by migration of slag leachate through ground water that discharges to surface water or direct overland (stormwater) run-off of dissolved or suspended slag materials. The constituent concentrations and pH levels detected in blast furnace and steel furnace slag leachate confirm that the potential exists for slag contaminants to migrate into surface water in a leached form. The potential for overland release of slag particles to surface waters is limited considerably by the generally large size of the slag fragments. A small fraction of the slag particles that are 0.1 mm or less in size tend to be appreciably erodible[3], and only a very small fraction of the blast furnace and steel furnace slag solids are expected to be in this size range.
>
> Based on environmental settings of the facilities and the presence of stormwater run-on/run-off controls at slag management units, the potential for contaminants from blast furnace and steel furnace slag to migrate into surface water at the eleven facilities appears to range from relatively low to relatively high. The potential for significant exposure to these contaminants, however, appears moderate at most.

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[3] "As indicated by the soil erodibility factor of the USDA's Universal Soil Loss Equation."

D.R. de Villiers of Monash University in Clayton, Australia studied the leaching of arsenic-doped slags for his doctoral dissertation (deV95). While the primary focus was on As leaching, he also developed some quantitative data on Fe, Mn, and Pb and qualitative information on other elements. The studies involved four commercial slags from lead-zinc smelters and two synthetic slags. To obtain the desired As levels, the commercial slags were remelted with appropriate As additions at 1,300 to 1,400°C in an electric muffle furnace. Slags 1-4 were produced from commercial slags A-D with a nominal As content of 0.66%, slags 5-8 were produced from commercial slags A-D with a nominal As content of 2.66%, and slags 9 and 10 were prepared in the laboratory by blending and melting the requisite raw materials. Nominal compositions for the six base slags are listed in Table I-5.

Table I-5. Nominal Compositions (wt%) of Slag Mixtures Studied by de Villiers

Component	Slag A	Slag B	Slag C	Slag D	Slag 9	Slag 10
"FeO"	41.0	27.9	37.2	30.3	32.2	33.4
SiO_2	19.5	21.7	25.8	23.9	22.4	23.2
CaO	19.0	15.1	19.1	17.9	28.7	29.8
ZnO	7.5	22.2	3.5	16.7		
Al_2O_3	7.0	5.6	9.1	4.7	7.9	8.2
Pb	0.5	2.0	0.023	2.35		
S	2.0	2.1	1.4	0.93		
MnO		3.4	4.8		5.2	5.4
MgO	2.0	1.3	1.5	1.5		
Cu	0.4	0.18	0.68	0.16		

Note: Slag A was used to produce Slags 1 & 5, Slag B was used to produce slags 2 & 6, etc.

Slags were leached for up to 40 weeks using either the EPA TCLP or SPLP[4] leaching procedures. Temporal variation in the concentration of elements in the SPLP leach solutions from Slags 1 and 3 is presented in Table I-6.

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[4] The SPLP procedure uses a very dilute solution of sulfuric acid and nitric acid in water as the extractant to simulate acid rain. Since the solution is not buffered the pH is subject to change during the leaching process.

Table I-6. Variation in the Concentration of Elements Leached from Slags 1 and 3 in SPLP Solutions (deV95)

Observed Behavior	Slag 1	Slag 3
Increase with time	Ca, Mn, Sr, Ba, Ti	Ca, Mn, Sr, Ba, Ti
Decrease with time	Fe, Cu, Zn, As, Pb,	Fe, Cu, Zn, As, Pb, Al
Similar concentration	Al, Sb	Sb
pH (18 hr)/pH (40 weeks)	7.9/8.6	8.7/7.5

In contrast to the Corps of Engineers data presented above, where the concentration of Ca, Ba, and Mn in the leachate was independent of time, de Villiers found these elements increased with time. A comparison of Corps of Engineers leaching data with those of de Villiers for a roughly comparable time period is shown in Table I-7.

Table I-7. Comparison of Corps of Engineers and de Villiers Leaching Data[1]

Source	Slag	Leachate	pH	Fe (ppb)	Mn (ppb)	Pb (ppb)
deV95	1	SPLP sol'n	7.1	~0	73	~0
deV95	2	SPLP sol'n	6.9	~0	214	~0
deV95	3	SPLP sol'n	7.4	~0	150	~0
deV95	4	SPLP sol'n	6.9	~0	16	~0
USA89	2	Nat'l precip.	8.4	<100	<10	<2

1 - DeV95-4 weeks, USA89-42 days leaching

The results on Fe, Pb and Mn leaching from the two studies indicate similar concentrations. The higher Mn levels observed by de Villiers may be related to the leach solution pH. His studies indicated that when the pH of the extractant for Slag 6 (same source as Slag 2) was 10.2, the Mn concentration was 20 ppb.

Dehmel et al. conducted Ra leaching tests on finely ground slags in de-ionized water, 6N HCl, and 4N HNO_3 (DEH92). In each test, one gram of slag was mixed with 500 ml of solution and stirred for 24 hours. In these aggressive tests, all the radium was solubilized in the acid solutions and 8% of the Ra was solubilized in the de-ionized water.

I.3 POSSIBLE MODELING APPROACH

Unfortunately, it is difficult to use the limited data described above for modeling leaching of radionuclides from slag piles. Given this caveat, the following recommendations are made for interim modeling:

Constant source term approach - *Use mean values for Weathering Bed Leachates in Table I-2 for Ba, Ca, Cr, Fe, K, Mn, Se (use P data for Se), and Sr (use Ba data for Sr). Use Cr data for other strong oxide formers (e.g., Ac, Am, Ce, Cm, Eu, Nb, Np, Pa, Pm, Pu, Ra, Sm, Th, U, Y, and Zr).*

Time varying source term approach - *Use equation 6 for Sr, Ca and Ba. Assume Cs leaches 10 times as fast as Sr. Use equation 7 for Co, Fe, Mn, and Ni. Use equation 10 for Cr and other oxide formers (e.g., Ac, Am, Ce, Cm, Eu, Nb, Np, Pa, Pm, Pu, Ra, Sm, Th, U, Y, and Zr).*

Use of the data obtained from slag cement leaching studies is believed to be conservative since the radionuclides in the cement composites are not in dissolved in the slag and therefore not expected to be as tightly bound in the solid matrix.

REFERENCES

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APPENDIX I-2

PRELIMINARY RESULTS OF LEACH RATE STUDY

performed by

BROOKHAVEN NATIONAL LABORATORY

Brookhaven National Laboratory
MEMORANDUM

Date: February 3, 1997
To: Carey Johnston, EPA
From: M. Fuhrmann
Subject: Leach Rates of Slags

We have determined that releases of Sr generally can be described by diffusion. For the AS-3 column experiment Incremental fraction releases vs Time follows the equation $IFR = 0.0075t^{-1/2}$ which indicates diffusion control. Examining the ALT data we find that diffusion coefficients for the AS and E series monolithic samples are:

AS-1 = 1.4×10^{-11} cm^2/S	E-1 = 8.5×10^{-11}
AS-2 = 2.5×10^{-11}	E-2 = Linear release at 8.3×10^{4} /day
AS-3 = 6.2×10^{-12}	E-3 = 5.5×10^{-11}

Assuming a cylinder of 1 cm height and 1 cm diameter, we have calculated the cumulative fractional release (CFR) for Sr at various times, with a diffusion coefficient of 2.5×10^{11}. Results are:

1 year	CFR = 0. 178
10 years	CFR = 0.495
20 years	CFR = 0.642
100 years	CFR = 0.958

From the AS-3 column data we have determined that releases of Si are not diffusion controlled and speculate that releases are related to solubility in the alkaline leachate. This requires an induction period during which Si concentrations in the leachate increase. After about 20 days they become more linear but with a lot of scatter. The average rate is 3.85×10^{5} fraction/day. Based on this linear rate about 1.4 % of the original Si would be released in one year.

Al in the column effluent and in the leachate from the monolithic samples appears to be diffusion controlled. Diffusion coefficients from the ALT experiments are:

AS-1 = 3.4×10^{-15} cm^2/S   E-1 = 3.7×10^{-10}
AS-2 = 2.8×10^{-10} E-2 = 3.1×10^{-11}
AS-3 = 8.5×10^{-11} E-3 = 7.2×10^{-13}

Using the diffusion coefficient from ALT sample AS-3, we estimate releases of a 1 cm x 1 cm cylinder as:

1 year   CFL = 0.314
10 years   CFL = 0.762
55 years   CFL = 0.999

APPENDIX J

NORMALIZED DOSES AND RISKS TO MAXIMALLY EXPOSED INDIVIDUALS - BY SCENARIO

Table of Contents

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SCRDRIVE: Driver, inside cab of vehicle

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	5.80E-03	4.41E-09
Fe-55	0.00E+00	0.00E+00
Co-60	1.81E-02	1.38E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	4.16E-03	3.16E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	1.08E-02	8.23E-09
Mo-93	1.17E-07	8.88E-14
Tc-99	4.32E-10	3.28E-16
Ru-106+D	1.35E-03	1.03E-09
Ag-110m	1.90E-02	1.45E-08
Sb-125	2.57E-03	1.96E-09
I-129	1.94E-06	1.48E-12
Cs-134	1.05E-02	7.95E-09
Cs-137+D	3.74E-03	2.85E-09
Ce-144+D	2.56E-04	1.94E-10
Pm-147	5.51E-09	4.19E-15
Eu-152	7.54E-03	5.73E-09
Pb-210+D	4.37E-07	3.32E-13
Ra-226+D	1.19E-02	9.08E-09
Ra-228+D	6.24E-03	4.75E-09
Ac-227+D	1.71E-03	1.30E-09
Th-228+D	1.03E-02	7.85E-09
Th-229+D	1.39E-03	1.05E-09
Th-230	4.37E-07	3.32E-13
Th-232	1.31E-07	9.95E-14
Pa-231	1.30E-04	9.87E-11
U-234	1.02E-07	7.73E-14
U-235+D	4.42E-04	3.36E-10
U-238+D	1.28E-04	9.75E-11
Np-237+D	9.36E-04	7.12E-10
Pu-238	1.64E-08	1.25E-14
Pu-239	7.80E-08	5.93E-14
Pu-240	1.63E-08	1.24E-14
Pu-241+D	6.84E-09	5.20E-15
Pu-242	1.45E-08	1.10E-14
Am-241	7.28E-06	5.54E-12
Cm-244	1.38E-08	1.05E-14
U-Series	1.22E-02	9.26E-09
U-Separ.	1.49E-04	1.13E-10
U-Deplete	1.35E-04	1.03E-10
Th-Series	1.66E-02	1.26E-08

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SCRAPCUT: Cutting/sizing scrap for furnace charge

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	3.48E-07	2.65E-13
Mn-54	1.34E-01	1.02E-07
Fe-55	0.00E+00	0.00E+00
Co-60	4.21E-01	3.20E-07
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	9.60E-02	7.30E-08
Sr-90+D	0.00E+00	0.00E+00
Nb-94	2.50E-01	1.90E-07
Mo-93	1.53E-05	1.17E-11
Tc-99	3.25E-06	2.47E-12
Ru-106+D	3.34E-02	2.54E-08
Ag-110m	4.46E-01	3.39E-07
Sb-125	6.37E-02	4.85E-08
I-129	3.37E-04	2.56E-10
Cs-134	2.46E-01	1.87E-07
Cs-137+D	8.87E-02	6.75E-08
Ce-144+D	8.42E-03	6.40E-09
Pm-147	1.30E-06	9.90E-13
Eu-152	1.82E-01	1.38E-07
Pb-210+D	1.58E-04	1.20E-10
Ra-226+D	2.91E-01	2.21E-07
Ra-228+D	1.55E-01	1.18E-07
Ac-227+D	5.23E-02	3.98E-08
Th-228+D	2.64E-01	2.01E-07
Th-229+D	4.14E-02	3.15E-08
Th-230	3.14E-05	2.39E-11
Th-232	1.35E-05	1.03E-11
Pa-231	4.96E-03	3.77E-09
U-234	1.04E-05	7.93E-12
U-235+D	1.97E-02	1.50E-08
U-238+D	4.03E-03	3.06E-09
Np-237+D	2.84E-02	2.16E-08
Pu-238	3.94E-06	2.99E-12
Pu-239	7.64E-06	5.81E-12
Pu-240	3.80E-06	2.89E-12
Pu-241+D	4.91E-07	3.74E-13
Pu-242	3.32E-06	2.53E-12
Am-241	1.13E-03	8.62E-10
Cm-244	3.28E-06	2.49E-12
U-Series	2.99E-01	2.27E-07
U-Separ.	4.96E-03	3.77E-09
U-Deplete	4.34E-03	3.30E-09
Th-Series	4.19E-01	3.19E-07

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SCRAPCUT: Cutting/sizing scrap for furnace charge

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	2.44E-06	8.18E-15	1.22E-05	6.04E-12
Mn-54	7.84E-06	4.32E-12	1.62E-05	1.15E-11
Fe-55	3.14E-06	6.54E-13	3.55E-06	2.06E-12
Co-60	2.56E-04	8.05E-11	1.58E-04	1.11E-10
Ni-59	3.16E-06	4.68E-13	1.23E-06	1.08E-12
Ni-63	7.36E-06	1.19E-12	3.38E-06	3.23E-12
Zn-65	2.39E-05	1.17E-11	8.44E-05	5.80E-11
Sr-90+D	1.53E-03	8.13E-11	8.96E-04	3.26E-10
Nb-94	4.85E-04	9.61E-11	4.18E-05	4.05E-11
Mo-93	3.33E-05	0.00E+00	7.88E-06	0.00E+00
Tc-99	9.74E-06	3.38E-12	8.55E-06	8.20E-12
Ru-106+D	5.58E-04	1.35E-10	1.60E-04	2.02E-10
Ag-110m	9.39E-05	3.76E-11	6.32E-05	4.94E-11
Sb-125	1.43E-05	6.86E-12	1.64E-05	2.07E-11
I-129	2.03E-04	1.43E-10	1.61E-03	1.08E-09
Cs-134	5.41E-05	3.38E-11	4.29E-04	2.77E-10
Cs-137+D	3.74E-05	2.24E-11	2.92E-04	1.85E-10
Ce-144+D	4.37E-04	1.26E-10	1.24E-04	1.73E-10
Pm-147	4.59E-05	8.74E-12	6.12E-06	8.27E-12
Eu-152	2.58E-04	9.26E-11	3.79E-05	3.35E-11
Pb-210+D	2.71E-02	4.52E-09	4.25E-02	5.89E-09
Ra-226+D	1.01E-02	3.21E-09	7.76E-03	1.73E-09
Ra-228+D	5.94E-03	1.16E-09	8.41E-03	1.45E-09
Ac-227+D	7.86E+00	9.19E-08	8.63E-02	3.67E-09
Th-228+D	4.04E-01	1.13E-07	4.73E-03	1.35E-09
Th-229+D	2.53E+00	9.64E-08	2.36E-02	2.09E-09
Th-230	3.81E-01	2.02E-08	3.20E-03	2.19E-10
Th-232	1.92E+00	2.26E-08	1.60E-02	1.92E-10
Pa-231	1.50E+00	2.83E-08	6.19E-02	8.70E-10
U-234	1.55E-01	1.63E-08	1.66E-03	2.60E-10
U-235+D	1.44E-01	1.52E-08	1.56E-03	2.75E-10
U-238+D	1.39E-01	1.46E-08	1.57E-03	3.62E-10
Np-237+D	6.32E-01	4.04E-08	2.60E-02	1.75E-09
Pu-238	4.59E-01	3.21E-08	1.87E-02	1.73E-09
Pu-239	5.02E-01	3.25E-08	2.07E-02	1.85E-09
Pu-240	5.02E-01	3.25E-08	2.07E-02	1.84E-09
Pu-241+D	9.65E-03	3.29E-10	4.00E-04	3.03E-11
Pu-242	4.81E-01	3.09E-08	1.97E-02	1.76E-09
Am-241	5.19E-01	4.50E-08	2.13E-02	1.92E-09
Cm-244	2.90E-01	2.85E-08	1.18E-02	1.23E-09
U-Series	1.16E+00	6.52E-08	6.38E-02	8.68E-09
U-Separ.	3.00E-01	3.16E-08	3.30E-03	6.34E-10
U-Deplete	1.55E-01	1.63E-08	1.75E-03	3.90E-10
Th-Series	2.33E+00	1.37E-07	2.91E-02	2.99E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation OP-CRANE: Moving scrap by rectangular charging bucket

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	6.85E-03	5.21E-09
Fe-55	0.00E+00	0.00E+00
Co-60	2.18E-02	1.65E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	5.01E-03	3.81E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	1.28E-02	9.72E-09
Mo-93	4.70E-08	3.57E-14
Tc-99	4.41E-10	3.35E-16
Ru-106+D	1.59E-03	1.21E-09
Ag-110m	2.27E-02	1.73E-08
Sb-125	3.00E-03	2.28E-09
I-129	1.67E-06	1.27E-12
Cs-134	1.23E-02	9.37E-09
Cs-137+D	4.39E-03	3.34E-09
Ce-144+D	3.06E-04	2.33E-10
Pm-147	5.81E-09	4.42E-15
Eu-152	8.55E-03	6.50E-09
Pb-210+D	4.16E-07	3.16E-13
Ra-226+D	1.40E-02	1.07E-08
Ra-228+D	7.06E-03	5.37E-09
Ac-227+D	2.04E-03	1.55E-09
Th-228+D	1.20E-02	9.14E-09
Th-229+D	1.36E-03	1.04E-09
Th-230	4.58E-07	3.48E-13
Th-232	1.31E-07	9.96E-14
Pa-231	1.46E-04	1.11E-10
U-234	1.00E-07	7.61E-14
U-235+D	4.81E-04	3.66E-10
U-238+D	1.10E-04	8.37E-11
Np-237+D	1.05E-03	8.02E-10
Pu-238	9.43E-09	7.17E-15
Pu-239	7.85E-08	5.97E-14
Pu-240	9.66E-09	7.35E-15
Pu-241+D	7.43E-09	5.65E-15
Pu-242	8.97E-09	6.82E-15
Am-241	7.03E-06	5.34E-12
Cm-244	6.97E-09	5.30E-15
U-Series	1.43E-02	1.08E-08
U-Separ.	1.33E-04	1.01E-10
U-Deplete	1.18E-04	8.96E-11
Th-Series	1.91E-02	1.45E-08

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation OP-CRANE: Moving scrap by rectangular charging bucket

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	2.58E-06	1.42E-12	3.53E-05	2.50E-11
Fe-55	1.56E-07	3.24E-14	2.35E-06	1.36E-12
Co-60	2.55E-05	8.03E-12	3.97E-05	2.78E-11
Ni-59	1.07E-07	1.58E-14	8.12E-07	7.16E-13
Ni-63	2.69E-07	4.33E-14	2.23E-06	2.13E-12
Zn-65	2.25E-04	1.10E-10	5.27E-03	3.62E-09
Sr-90+D	1.12E-04	5.97E-12	2.30E-03	8.36E-10
Nb-94	1.88E-04	3.72E-11	1.07E-04	1.04E-10
Mo-93	3.32E-06	0.00E+00	9.33E-07	0.00E+00
Tc-99	9.71E-07	3.37E-13	5.66E-06	5.42E-12
Ru-106+D	5.57E-05	1.34E-11	1.06E-04	1.33E-10
Ag-110m	1.14E-04	4.56E-11	1.03E-03	8.04E-10
Sb-125	2.81E-05	1.35E-11	2.14E-04	2.70E-10
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	5.10E-04	3.18E-10	2.68E-02	1.73E-08
Cs-137+D	3.52E-04	2.11E-10	1.83E-02	1.16E-08
Ce-144+D	1.69E-04	4.88E-11	3.17E-04	4.45E-10
Pm-147	1.78E-05	3.38E-12	1.57E-05	2.12E-11
Eu-152	1.00E-04	3.59E-11	9.72E-05	8.61E-11
Pb-210+D	2.47E-01	4.11E-08	2.66E+00	3.68E-07
Ra-226+D	3.89E-03	1.24E-09	1.99E-02	4.44E-09
Ra-228+D	2.22E-03	4.34E-10	2.16E-02	3.71E-09
Ac-227+D	5.96E-01	6.96E-09	2.22E-01	9.41E-09
Th-228+D	1.56E-01	4.38E-08	1.21E-02	3.47E-09
Th-229+D	7.90E-01	3.01E-08	6.05E-02	5.36E-09
Th-230	1.18E-01	6.27E-09	8.22E-03	5.61E-10
Th-232	5.21E-01	6.13E-09	4.10E-02	4.92E-10
Pa-231	3.89E-01	7.33E-09	1.59E-01	2.23E-09
U-234	6.00E-02	6.32E-09	3.92E-04	6.14E-11
U-235+D	5.56E-02	5.88E-09	4.21E-04	7.39E-11
U-238+D	5.36E-02	5.64E-09	5.62E-04	1.29E-10
Np-237+D	2.45E-01	1.56E-08	6.67E-02	4.50E-09
Pu-238	1.31E-01	9.13E-09	7.44E-04	6.87E-11
Pu-239	1.40E-01	9.04E-09	7.78E-04	6.94E-11
Pu-240	1.40E-01	9.04E-09	7.78E-04	6.93E-11
Pu-241+D	2.25E-03	7.64E-11	1.15E-05	8.70E-13
Pu-242	1.33E-01	8.52E-09	7.39E-04	6.60E-11
Am-241	2.01E-01	1.74E-08	5.47E-02	4.93E-09
Cm-244	1.12E-01	1.10E-08	3.03E-02	3.16E-09
U-Series	5.31E-01	6.15E-08	2.71E+00	3.73E-07
U-Separ.	1.16E-01	1.22E-08	9.74E-04	1.94E-10
U-Deplete	6.00E-02	6.31E-09	6.04E-04	1.36E-10
Th-Series	6.80E-01	5.03E-08	7.47E-02	7.67E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation FURNACE: Exposure from EAF during melt

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	8.82E-05	6.71E-11
Fe-55	0.00E+00	0.00E+00
Co-60	6.54E-04	4.98E-10
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	1.16E-04	8.81E-11
Sr-90+D	0.00E+00	0.00E+00
Nb-94	1.50E-04	1.14E-10
Mo-93	4.70E-29	0.00E+00
Tc-99	1.69E-21	1.29E-27
Ru-106+D	1.39E-05	1.05E-11
Ag-110m	4.04E-04	3.07E-10
Sb-125	1.31E-05	9.99E-12
I-129	1.02E-28	0.00E+00
Cs-134	1.21E-04	9.19E-11
Cs-137+D	3.23E-05	2.46E-11
Ce-144+D	1.33E-05	1.01E-11
Pm-147	3.68E-16	2.80E-22
Eu-152	2.04E-04	1.55E-10
Pb-210+D	8.16E-10	6.21E-16
Ra-226+D	4.81E-04	3.66E-10
Ra-228+D	1.49E-04	1.13E-10
Ac-227+D	4.60E-06	3.50E-12
Th-228+D	8.03E-04	6.11E-10
Th-229+D	1.68E-05	1.28E-11
Th-230	2.87E-12	2.18E-18
Th-232	1.10E-14	8.33E-21
Pa-231	7.31E-08	5.56E-14
U-234	5.35E-15	4.07E-21
U-235+D	1.05E-08	7.98E-15
U-238+D	2.48E-06	1.89E-12
Np-237+D	6.83E-07	5.20E-13
Pu-238	5.45E-30	0.00E+00
Pu-239	9.74E-16	7.41E-22
Pu-240	5.20E-30	0.00E+00
Pu-241+D	8.62E-13	6.56E-19
Pu-242	4.31E-30	0.00E+00
Am-241	4.43E-27	0.00E+00
Cm-244	5.35E-30	0.00E+00
U-Series	4.84E-04	3.68E-10
U-Separ.	2.48E-06	1.89E-12
U-Deplete	2.48E-06	1.89E-12
Th-Series	9.51E-04	7.24E-10

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation FURNACE: Exposure from EAF during melt

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	4.36E-06	2.41E-12	3.53E-05	2.50E-11
Fe-55	2.64E-07	5.49E-14	2.35E-06	1.36E-12
Co-60	4.32E-05	1.36E-11	3.97E-05	2.78E-11
Ni-59	1.81E-07	2.68E-14	8.12E-07	7.16E-13
Ni-63	4.54E-07	7.33E-14	2.23E-06	2.13E-12
Zn-65	3.80E-04	1.86E-10	5.27E-03	3.62E-09
Sr-90+D	1.90E-04	1.01E-11	2.30E-03	8.36E-10
Nb-94	3.18E-04	6.29E-11	1.07E-04	1.04E-10
Mo-93	5.61E-06	0.00E+00	9.33E-07	0.00E+00
Tc-99	1.64E-06	5.71E-13	5.66E-06	5.42E-12
Ru-106+D	9.43E-05	2.27E-11	1.06E-04	1.33E-10
Ag-110m	1.93E-04	7.71E-11	1.03E-03	8.04E-10
Sb-125	4.75E-05	2.28E-11	2.14E-04	2.70E-10
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	8.63E-04	5.38E-10	2.68E-02	1.73E-08
Cs-137+D	5.96E-04	3.58E-10	1.83E-02	1.16E-08
Ce-144+D	2.86E-04	8.26E-11	3.17E-04	4.45E-10
Pm-147	3.01E-05	5.73E-12	1.57E-05	2.12E-11
Eu-152	1.69E-04	6.07E-11	9.72E-05	8.61E-11
Pb-210+D	4.17E-01	6.95E-08	2.66E+00	3.68E-07
Ra-226+D	6.59E-03	2.10E-09	1.99E-02	4.44E-09
Ra-228+D	3.75E-03	7.35E-10	2.16E-02	3.71E-09
Ac-227+D	1.01E+00	1.18E-08	2.22E-01	9.41E-09
Th-228+D	2.64E-01	7.41E-08	1.21E-02	3.47E-09
Th-229+D	1.34E+00	5.09E-08	6.05E-02	5.36E-09
Th-230	2.00E-01	1.06E-08	8.22E-03	5.61E-10
Th-232	8.82E-01	1.04E-08	4.10E-02	4.92E-10
Pa-231	6.58E-01	1.24E-08	1.59E-01	2.23E-09
U-234	1.02E-01	1.07E-08	3.92E-04	6.14E-11
U-235+D	9.41E-02	9.96E-09	4.21E-04	7.39E-11
U-238+D	9.07E-02	9.54E-09	5.62E-04	1.29E-10
Np-237+D	4.14E-01	2.65E-08	6.67E-02	4.50E-09
Pu-238	2.21E-01	1.55E-08	7.44E-04	6.87E-11
Pu-239	2.36E-01	1.53E-08	7.78E-04	6.94E-11
Pu-240	2.36E-01	1.53E-08	7.78E-04	6.93E-11
Pu-241+D	3.80E-03	1.29E-10	1.15E-05	8.70E-13
Pu-242	2.25E-01	1.44E-08	7.39E-04	6.60E-11
Am-241	3.40E-01	2.95E-08	5.47E-02	4.93E-09
Cm-244	1.90E-01	1.87E-08	3.03E-02	3.16E-09
U-Series	8.99E-01	1.04E-07	2.71E+00	3.73E-07
U-Separ.	1.97E-01	2.07E-08	9.74E-04	1.94E-10
U-Deplete	1.02E-01	1.07E-08	6.04E-04	1.36E-10
Th-Series	1.15E+00	8.52E-08	7.47E-02	7.67E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation OPCASTER: Exposure from continuous caster

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	1.19E-02	9.04E-09
Fe-55	0.00E+00	0.00E+00
Co-60	5.75E-02	4.38E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	2.57E-03	1.96E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	5.09E-09	3.87E-15
Tc-99	1.25E-09	9.48E-16
Ru-106+D	4.18E-03	3.18E-09
Ag-110m	5.97E-02	4.54E-08
Sb-125	7.93E-03	6.03E-09
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation TUNDISH: Exposure from molten steel in tundish

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	1.50E-03	1.14E-09
Fe-55	0.00E+00	0.00E+00
Co-60	9.54E-03	7.26E-09
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	3.93E-04	2.99E-10
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	7.83E-29	0.00E+00
Tc-99	8.76E-15	6.66E-21
Ru-106+D	4.24E-04	3.23E-10
Ag-110m	7.97E-03	6.06E-09
Sb-125	6.62E-04	5.03E-10
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation OPCASTER: Exposure from continuous caster

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	3.97E-06	2.19E-12	3.53E-05	2.50E-11
Fe-55	2.40E-07	4.99E-14	2.35E-06	1.36E-12
Co-60	3.93E-05	1.24E-11	3.97E-05	2.78E-11
Ni-59	1.65E-07	2.43E-14	8.12E-07	7.16E-13
Ni-63	4.13E-07	6.66E-14	2.23E-06	2.13E-12
Zn-65	3.46E-04	1.69E-10	5.27E-03	3.62E-09
Sr-90+D	1.73E-04	9.18E-12	2.30E-03	8.36E-10
Nb-94	2.89E-04	5.72E-11	1.07E-04	1.04E-10
Mo-93	5.10E-06	0.00E+00	9.33E-07	0.00E+00
Tc-99	1.49E-06	5.19E-13	5.66E-06	5.42E-12
Ru-106+D	8.57E-05	2.07E-11	1.06E-04	1.33E-10
Ag-110m	1.75E-04	7.01E-11	1.03E-03	8.04E-10
Sb-125	4.32E-05	2.08E-11	2.14E-04	2.70E-10
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	7.84E-04	4.90E-10	2.68E-02	1.73E-08
Cs-137+D	5.42E-04	3.25E-10	1.83E-02	1.16E-08
Ce-144+D	2.60E-04	7.51E-11	3.17E-04	4.45E-10
Pm-147	2.73E-05	5.21E-12	1.57E-05	2.12E-11
Eu-152	1.54E-04	5.52E-11	9.72E-05	8.61E-11
Pb-210+D	3.79E-01	6.32E-08	2.66E+00	3.68E-07
Ra-226+D	5.99E-03	1.91E-09	1.99E-02	4.44E-09
Ra-228+D	3.41E-03	6.68E-10	2.16E-02	3.71E-09
Ac-227+D	9.16E-01	1.07E-08	2.22E-01	9.41E-09
Th-228+D	2.40E-01	6.73E-08	1.21E-02	3.47E-09
Th-229+D	1.22E+00	4.63E-08	6.05E-02	5.36E-09
Th-230	1.82E-01	9.65E-09	8.22E-03	5.61E-10
Th-232	8.02E-01	9.43E-09	4.10E-02	4.92E-10
Pa-231	5.98E-01	1.13E-08	1.59E-01	2.23E-09
U-234	9.23E-02	9.72E-09	3.92E-04	6.14E-11
U-235+D	8.56E-02	9.05E-09	4.21E-04	7.39E-11
U-238+D	8.25E-02	8.67E-09	5.62E-04	1.29E-10
Np-237+D	3.76E-01	2.41E-08	6.67E-02	4.50E-09
Pu-238	2.01E-01	1.41E-08	7.44E-04	6.87E-11
Pu-239	2.15E-01	1.39E-08	7.78E-04	6.94E-11
Pu-240	2.15E-01	1.39E-08	7.78E-04	6.93E-11
Pu-241+D	3.45E-03	1.18E-10	1.15E-05	8.70E-13
Pu-242	2.04E-01	1.31E-08	7.39E-04	6.60E-11
Am-241	3.09E-01	2.68E-08	5.47E-02	4.93E-09
Cm-244	1.73E-01	1.70E-08	3.03E-02	3.16E-09
U-Series	8.18E-01	9.46E-08	2.71E+00	3.73E-07
U-Separ.	1.79E-01	1.88E-08	9.74E-04	1.94E-10
U-Deplete	9.23E-02	9.70E-09	6.04E-04	1.36E-10
Th-Series	1.05E+00	7.75E-08	7.47E-02	7.67E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation BAGHOUSE: Handling the bag house filters

Pathway:	External		Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	2.72E-04	2.07E-10	2.97E-07	1.64E-13	8.07E-07	5.72E-13
Fe-55	0.00E+00	0.00E+00	1.80E-08	3.74E-15	5.37E-08	3.11E-14
Co-60	2.27E-04	1.72E-10	2.94E-06	9.25E-13	9.07E-07	6.36E-13
Ni-59	0.00E+00	0.00E+00	1.23E-08	1.82E-15	1.86E-08	1.64E-14
Ni-63	0.00E+00	0.00E+00	3.09E-08	4.99E-15	5.10E-08	4.88E-14
Zn-65	8.05E-03	6.12E-09	2.59E-05	1.27E-11	1.21E-04	8.29E-11
Sr-90+D	0.00E+00	0.00E+00	1.29E-05	6.87E-13	5.26E-05	1.91E-11
Nb-94	6.10E-04	4.64E-10	2.16E-05	4.28E-12	2.45E-06	2.37E-12
Mo-93	8.87E-08	6.75E-14	3.82E-07	0.00E+00	2.13E-08	0.00E+00
Tc-99	5.57E-11	4.24E-17	1.12E-07	3.88E-14	1.29E-07	1.24E-13
Ru-106+D	2.13E-05	1.62E-11	6.42E-06	1.55E-12	2.42E-06	3.05E-12
Ag-110m	6.52E-03	4.96E-09	1.31E-05	5.25E-12	2.35E-05	1.84E-11
Sb-125	8.83E-04	6.71E-10	3.23E-06	1.55E-12	4.88E-06	6.17E-12
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	2.24E-02	1.70E-08	5.87E-05	3.67E-11	6.12E-04	3.96E-10
Cs-137+D	8.19E-03	6.23E-09	4.06E-05	2.43E-11	4.17E-04	2.64E-10
Ce-144+D	1.82E-05	1.38E-11	1.95E-05	5.62E-12	7.25E-06	1.02E-11
Pm-147	1.45E-09	1.11E-15	2.05E-06	3.90E-13	3.59E-07	4.85E-13
Eu-152	4.30E-04	3.27E-10	1.15E-05	4.13E-12	2.22E-06	1.97E-12
Pb-210+D	1.73E-05	1.32E-11	2.84E-02	4.73E-09	6.08E-02	8.41E-09
Ra-226+D	6.43E-04	4.89E-10	4.49E-04	1.43E-10	4.55E-04	1.02E-10
Ra-228+D	3.48E-04	2.65E-10	2.56E-04	5.00E-11	4.94E-04	8.49E-11
Ac-227+D	1.83E-04	1.39E-10	6.86E-02	8.01E-10	5.07E-03	2.15E-10
Th-228+D	5.26E-04	4.00E-10	1.80E-02	5.04E-09	2.77E-04	7.93E-11
Th-229+D	1.40E-04	1.06E-10	9.10E-02	3.46E-09	1.38E-03	1.23E-10
Th-230	1.72E-07	1.31E-13	1.36E-02	7.23E-10	1.88E-04	1.28E-11
Th-232	8.39E-08	6.38E-14	6.00E-02	7.06E-10	9.37E-04	1.12E-11
Pa-231	1.27E-05	9.66E-12	4.48E-02	8.44E-10	3.63E-03	5.10E-11
U-234	6.74E-08	5.13E-14	6.91E-03	7.28E-10	8.96E-06	1.40E-12
U-235+D	6.76E-05	5.14E-11	6.41E-03	6.78E-10	9.63E-06	1.69E-12
U-238+D	8.01E-06	6.09E-12	6.18E-03	6.49E-10	1.28E-05	2.96E-12
Np-237+D	9.82E-05	7.47E-11	2.82E-02	1.80E-09	1.53E-03	1.03E-10
Pu-238	3.96E-08	3.01E-14	1.50E-02	1.05E-09	1.70E-05	1.57E-12
Pu-239	3.28E-08	2.50E-14	1.61E-02	1.04E-09	1.78E-05	1.59E-12
Pu-240	3.91E-08	2.98E-14	1.61E-02	1.04E-09	1.78E-05	1.58E-12
Pu-241+D	1.36E-09	1.04E-15	2.59E-04	8.80E-12	2.63E-07	1.99E-14
Pu-242	3.31E-08	2.51E-14	1.53E-02	9.82E-10	1.69E-05	1.51E-12
Am-241	9.11E-06	6.93E-12	2.32E-02	2.01E-09	1.25E-03	1.13E-10
Cm-244	3.52E-08	2.68E-14	1.29E-02	1.27E-09	6.92E-04	7.22E-11
U-Series	6.81E-04	5.18E-10	6.12E-02	7.09E-09	6.18E-02	8.54E-09
U-Separ.	1.13E-05	8.56E-12	1.34E-02	1.41E-09	2.23E-05	4.44E-12
U-Deplete	9.09E-06	6.92E-12	6.91E-03	7.27E-10	1.38E-05	3.11E-12
Th-Series	8.74E-04	6.65E-10	7.83E-02	5.80E-09	1.71E-03	1.75E-10

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation DST-TRK: Working under the bag house (from dust in the truck)

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	2.49E-03	1.89E-09
Fe-55	0.00E+00	0.00E+00
Co-60	2.36E-03	1.79E-09
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	5.12E-02	3.89E-08
Sr-90+D	0.00E+00	0.00E+00
Nb-94	5.47E-03	4.16E-09
Mo-93	3.32E-12	2.52E-18
Tc-99	5.91E-11	4.50E-17
Ru-106+D	1.77E-04	1.35E-10
Ag-110m	6.13E-02	4.66E-08
Sb-125	6.68E-03	5.08E-09
I-129	0.00E+00	0.00E+00
Cs-134	1.29E-01	9.81E-08
Cs-137+D	4.63E-02	3.52E-08
Ce-144+D	1.32E-04	1.01E-10
Pm-147	3.10E-09	2.35E-15
Eu-152	3.84E-03	2.92E-09
Pb-210+D	4.18E-06	3.18E-12
Ra-226+D	6.10E-03	4.64E-09
Ra-228+D	3.17E-03	2.41E-09
Ac-227+D	9.01E-04	6.85E-10
Th-228+D	5.33E-03	4.05E-09
Th-229+D	7.22E-04	5.49E-10
Th-230	2.40E-07	1.82E-13
Th-232	6.84E-08	5.20E-14
Pa-231	6.87E-05	5.23E-11
U-234	5.13E-08	3.90E-14
U-235+D	2.44E-04	1.85E-10
U-238+D	6.51E-05	4.95E-11
Np-237+D	4.95E-04	3.76E-10
Pu-238	3.55E-09	2.70E-15
Pu-239	4.15E-08	3.16E-14
Pu-240	3.66E-09	2.78E-15
Pu-241+D	3.72E-09	2.83E-15
Pu-242	3.62E-09	2.75E-15
Am-241	3.67E-06	2.79E-12
Cm-244	2.39E-09	1.82E-15
U-Series	6.23E-03	4.74E-09
U-Separ.	7.66E-05	5.83E-11
U-Deplete	6.90E-05	5.25E-11
Th-Series	8.50E-03	6.47E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation BGHS-BAG: Working under the bag house (from dust in/on the bags)

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	2.24E-04	1.71E-10
Fe-55	0.00E+00	0.00E+00
Co-60	2.08E-04	1.58E-10
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	4.49E-03	3.42E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	4.94E-04	3.76E-10
Mo-93	1.63E-32	0.00E+00
Tc-99	4.75E-13	3.62E-19
Ru-106+D	1.60E-05	1.21E-11
Ag-110m	5.49E-03	4.18E-09
Sb-125	5.99E-04	4.56E-10
I-129	0.00E+00	0.00E+00
Cs-134	1.15E-02	8.71E-09
Cs-137+D	4.13E-03	3.14E-09
Ce-144+D	1.07E-05	8.10E-12
Pm-147	1.07E-10	8.10E-17
Eu-152	3.38E-04	2.57E-10
Pb-210+D	6.53E-08	4.97E-14
Ra-226+D	5.32E-04	4.05E-10
Ra-228+D	2.81E-04	2.14E-10
Ac-227+D	7.51E-05	5.71E-11
Th-228+D	4.48E-04	3.41E-10
Th-229+D	5.95E-05	4.52E-11
Th-230	1.12E-08	8.48E-15
Th-232	1.86E-09	1.41E-15
Pa-231	5.71E-06	4.34E-12
U-234	1.51E-09	1.15E-15
U-235+D	1.59E-05	1.21E-11
U-238+D	5.59E-06	4.25E-12
Np-237+D	4.03E-05	3.06E-11
Pu-238	7.51E-15	5.71E-21
Pu-239	1.14E-09	8.65E-16
Pu-240	4.28E-15	3.26E-21
Pu-241+D	2.27E-10	1.73E-16
Pu-242	1.46E-14	1.11E-20
Am-241	7.94E-11	6.04E-17
Cm-244	1.20E-14	9.14E-21
U-Series	5.42E-04	4.12E-10
U-Separ.	6.34E-06	4.82E-12
U-Deplete	5.85E-06	4.45E-12
Th-Series	7.29E-04	5.55E-10

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation BGHS-FLR: Working under the bag house (from dust on the floor)

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	3.97E-04	3.02E-10
Fe-55	0.00E+00	0.00E+00
Co-60	3.67E-04	2.79E-10
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	9.92E-03	7.55E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	8.72E-04	6.63E-10
Mo-93	2.75E-32	0.00E+00
Tc-99	1.27E-12	9.68E-19
Ru-106+D	2.86E-05	2.17E-11
Ag-110m	9.66E-03	7.35E-09
Sb-125	1.08E-03	8.23E-10
I-129	0.00E+00	0.00E+00
Cs-134	2.56E-02	1.95E-08
Cs-137+D	9.24E-03	7.03E-09
Ce-144+D	1.89E-05	1.44E-11
Pm-147	2.37E-10	1.81E-16
Eu-152	5.98E-04	4.55E-10
Pb-210+D	1.15E-07	8.78E-14
Ra-226+D	9.37E-04	7.12E-10
Ra-228+D	4.98E-04	3.79E-10
Ac-227+D	1.39E-04	1.06E-10
Th-228+D	7.86E-04	5.98E-10
Th-229+D	1.09E-04	8.27E-11
Th-230	2.27E-08	1.73E-14
Th-232	4.11E-09	3.13E-15
Pa-231	1.07E-05	8.10E-12
U-234	3.36E-09	2.56E-15
U-235+D	3.20E-05	2.43E-11
U-238+D	9.87E-06	7.50E-12
Np-237+D	7.51E-05	5.71E-11
Pu-238	4.14E-14	3.15E-20
Pu-239	2.62E-09	1.99E-15
Pu-240	2.50E-14	1.90E-20
Pu-241+D	4.49E-10	3.42E-16
Pu-242	7.58E-14	5.76E-20
Am-241	3.37E-10	2.57E-16
Cm-244	6.08E-14	4.62E-20
U-Series	9.55E-04	7.27E-10
U-Separ.	1.14E-05	8.65E-12
U-Deplete	1.04E-05	7.89E-12
Th-Series	1.28E-03	9.77E-10

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation BAGHOUSE: Handling the bag house filters

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	3.80E-03	2.89E-09
Fe-55	0.00E+00	0.00E+00
Co-60	1.84E-02	1.40E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	8.29E-04	6.31E-10
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	3.97E-08	3.02E-14
Tc-99	3.73E-10	2.83E-16
Ru-106+D	1.34E-03	1.02E-09
Ag-110m	1.92E-02	1.46E-08
Sb-125	2.54E-03	1.93E-09
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation BGHS-IN: Bag house worker, inhalation & ingestion exposures

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	5.42E-06	2.99E-12	2.93E-05	2.07E-11
Fe-55	3.28E-07	6.82E-14	1.95E-06	1.13E-12
Co-60	5.37E-05	1.69E-11	3.29E-05	2.31E-11
Ni-59	2.25E-07	3.33E-14	6.73E-07	5.93E-13
Ni-63	5.65E-07	9.10E-14	1.85E-06	1.77E-12
Zn-65	4.73E-04	2.32E-10	4.37E-03	3.00E-09
Sr-90+D	2.36E-04	1.25E-11	1.91E-03	6.93E-10
Nb-94	3.95E-04	7.82E-11	8.88E-05	8.61E-11
Mo-93	6.98E-06	0.00E+00	7.73E-07	0.00E+00
Tc-99	2.04E-06	7.09E-13	4.69E-06	4.49E-12
Ru-106+D	1.17E-04	2.83E-11	8.78E-05	1.11E-10
Ag-110m	2.39E-04	9.58E-11	8.53E-04	6.66E-10
Sb-125	5.90E-05	2.84E-11	1.77E-04	2.24E-10
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	1.07E-03	6.69E-10	2.22E-02	1.43E-08
Cs-137+D	7.40E-04	4.45E-10	1.51E-02	9.57E-09
Ce-144+D	3.56E-04	1.03E-10	2.63E-04	3.69E-10
Pm-147	3.74E-05	7.12E-12	1.30E-05	1.76E-11
Eu-152	2.10E-04	7.54E-11	8.06E-05	7.13E-11
Pb-210+D	5.19E-01	8.64E-08	2.20E+00	3.05E-07
Ra-226+D	8.19E-03	2.61E-09	1.65E-02	3.68E-09
Ra-228+D	4.67E-03	9.13E-10	1.79E-02	3.08E-09
Ac-227+D	1.25E+00	1.46E-08	1.84E-01	7.80E-09
Th-228+D	3.29E-01	9.21E-08	1.01E-02	2.87E-09
Th-229+D	1.66E+00	6.32E-08	5.01E-02	4.44E-09
Th-230	2.49E-01	1.32E-08	6.81E-03	4.65E-10
Th-232	1.10E+00	1.29E-08	3.40E-02	4.07E-10
Pa-231	8.18E-01	1.54E-08	1.32E-01	1.85E-09
U-234	1.26E-01	1.33E-08	3.25E-04	5.09E-11
U-235+D	1.17E-01	1.24E-08	3.49E-04	6.13E-11
U-238+D	1.13E-01	1.19E-08	4.65E-04	1.07E-10
Np-237+D	5.15E-01	3.29E-08	5.53E-02	3.73E-09
Pu-238	2.75E-01	1.92E-08	6.17E-04	5.69E-11
Pu-239	2.94E-01	1.90E-08	6.44E-04	5.75E-11
Pu-240	2.94E-01	1.90E-08	6.44E-04	5.74E-11
Pu-241+D	4.72E-03	1.61E-10	9.53E-06	7.21E-13
Pu-242	2.79E-01	1.79E-08	6.12E-04	5.47E-11
Am-241	4.23E-01	3.67E-08	4.53E-02	4.08E-09
Cm-244	2.36E-01	2.32E-08	2.51E-02	2.62E-09
U-Series	1.12E+00	1.29E-07	2.24E+00	3.09E-07
U-Separ.	2.44E-01	2.57E-08	8.07E-04	1.61E-10
U-Deplete	1.26E-01	1.33E-08	5.01E-04	1.13E-10
Th-Series	1.43E+00	1.06E-07	6.19E-02	6.36E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation DUSTDRIV: Transporting bag house dust for disposal, cab of vehicle

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	9.48E-04	7.21E-10
Fe-55	0.00E+00	0.00E+00
Co-60	8.84E-04	6.72E-10
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	1.93E-02	1.47E-08
Sr-90+D	0.00E+00	0.00E+00
Nb-94	2.09E-03	1.59E-09
Mo-93	8.11E-09	6.17E-15
Tc-99	2.98E-11	2.27E-17
Ru-106+D	6.88E-05	5.23E-11
Ag-110m	2.33E-02	1.77E-08
Sb-125	2.63E-03	2.00E-09
I-129	0.00E+00	0.00E+00
Cs-134	4.96E-02	3.78E-08
Cs-137+D	1.79E-02	1.36E-08
Ce-144+D	5.06E-05	3.85E-11
Pm-147	1.43E-09	1.09E-15
Eu-152	1.46E-03	1.11E-09
Pb-210+D	3.10E-06	2.36E-12
Ra-226+D	2.30E-03	1.75E-09
Ra-228+D	1.20E-03	9.16E-10
Ac-227+D	3.67E-04	2.79E-10
Th-228+D	1.98E-03	1.50E-09
Th-229+D	2.88E-04	2.19E-10
Th-230	1.14E-07	8.66E-14
Th-232	3.58E-08	2.73E-14
Pa-231	2.82E-05	2.14E-11
U-234	2.73E-08	2.07E-14
U-235+D	1.06E-04	8.05E-11
U-238+D	2.51E-05	1.91E-11
Np-237+D	2.03E-04	1.55E-10
Pu-238	4.63E-09	3.52E-15
Pu-239	2.05E-08	1.56E-14
Pu-240	4.63E-09	3.52E-15
Pu-241+D	1.64E-09	1.25E-15
Pu-242	4.17E-09	3.17E-15
Am-241	2.28E-06	1.74E-12
Cm-244	3.81E-09	2.90E-15
U-Series	2.35E-03	1.79E-09
U-Separ.	3.01E-05	2.29E-11
U-Deplete	2.68E-05	2.04E-11
Th-Series	3.18E-03	2.42E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SLAGPILE: Slag pile at slag processor

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	1.99E-01	1.51E-07
Fe-55	0.00E+00	0.00E+00
Co-60	0.00E+00	0.00E+00
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	0.00E+00	0.00E+00
Sr-90+D	1.20E-03	9.12E-10
Nb-94	4.73E-01	3.60E-07
Mo-93	0.00E+00	0.00E+00
Tc-99	0.00E+00	0.00E+00
Ru-106+D	0.00E+00	0.00E+00
Ag-110m	0.00E+00	0.00E+00
Sb-125	0.00E+00	0.00E+00
I-129	0.00E+00	0.00E+00
Cs-134	2.44E-02	1.85E-08
Cs-137+D	8.79E-03	6.69E-09
Ce-144+D	1.59E-02	1.21E-08
Pm-147	2.45E-06	1.86E-12
Eu-152	3.43E-01	2.61E-07
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	5.47E-01	4.16E-07
Ra-228+D	2.92E-01	2.22E-07
Ac-227+D	9.85E-02	7.50E-08
Th-228+D	4.98E-01	3.79E-07
Th-229+D	7.81E-02	5.94E-08
Th-230	5.90E-05	4.49E-11
Th-232	2.54E-05	1.94E-11
Pa-231	9.34E-03	7.10E-09
U-234	1.96E-05	1.49E-11
U-235+D	3.70E-02	2.82E-08
U-238+D	7.58E-03	5.77E-09
Np-237+D	5.36E-02	4.08E-08
Pu-238	7.40E-06	5.62E-12
Pu-239	1.44E-05	1.09E-11
Pu-240	7.16E-06	5.45E-12
Pu-241+D	9.25E-07	7.04E-13
Pu-242	6.26E-06	4.76E-12
Am-241	2.13E-03	1.62E-09
Cm-244	6.15E-06	4.68E-12
U-Series	5.61E-01	4.27E-07
U-Separ.	9.34E-03	7.11E-09
U-Deplete	8.18E-03	6.22E-09
Th-Series	7.90E-01	6.01E-07

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SLAGPILE: Slag pile at slag processor

Pathway:	Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	1.25E-05	6.88E-12	9.83E-05	6.96E-11
Fe-55	7.69E-08	1.60E-14	6.66E-07	3.86E-13
Co-60	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sr-90+D	5.84E-04	3.11E-11	6.89E-03	2.50E-09
Nb-94	9.77E-04	1.94E-10	3.21E-04	3.11E-10
Mo-93	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Tc-99	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ru-106+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ag-110m	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sb-125	0.00E+00	0.00E+00	0.00E+00	0.00E+00
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	5.74E-06	3.58E-12	1.73E-04	1.12E-10
Cs-137+D	3.96E-06	2.38E-12	1.18E-04	7.48E-11
Ce-144+D	8.81E-04	2.54E-10	9.51E-04	1.33E-09
Pm-147	9.25E-05	1.76E-11	4.71E-05	6.36E-11
Eu-152	5.21E-04	1.87E-10	2.91E-04	2.58E-10
Pb-210+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-226+D	2.03E-02	6.47E-09	5.97E-02	1.33E-08
Ra-228+D	1.16E-02	2.26E-09	6.47E-02	1.11E-08
Ac-227+D	3.10E+00	3.62E-08	6.64E-01	2.82E-08
Th-228+D	8.13E-01	2.28E-07	3.64E-02	1.04E-08
Th-229+D	4.11E+00	1.57E-07	1.81E-01	1.61E-08
Th-230	6.17E-01	3.27E-08	2.46E-02	1.68E-09
Th-232	2.71E+00	3.19E-08	1.23E-01	1.47E-09
Pa-231	2.02E+00	3.82E-08	4.76E-01	6.69E-09
U-234	3.13E-01	3.29E-08	1.18E-03	1.84E-10
U-235+D	2.90E-01	3.06E-08	1.26E-03	2.22E-10
U-238+D	2.79E-01	2.94E-08	1.68E-03	3.88E-10
Np-237+D	1.27E+00	8.14E-08	2.00E-01	1.35E-08
Pu-238	6.80E-01	4.76E-08	2.23E-03	2.06E-10
Pu-239	7.27E-01	4.71E-08	2.33E-03	2.08E-10
Pu-240	7.27E-01	4.71E-08	2.33E-03	2.08E-10
Pu-241+D	1.17E-02	3.98E-10	3.45E-05	2.61E-12
Pu-242	6.91E-01	4.44E-08	2.21E-03	1.98E-10
Am-241	1.05E+00	9.08E-08	1.64E-01	1.48E-08
Cm-244	5.85E-01	5.74E-08	9.07E-02	9.47E-09
U-Series	1.48E+00	1.06E-07	1.41E-01	1.72E-08
U-Separ.	6.05E-01	6.37E-08	2.92E-03	5.82E-10
U-Deplete	3.12E-01	3.29E-08	1.81E-03	4.08E-10
Th-Series	3.54E+00	2.62E-07	2.24E-01	2.30E-08

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SLGLEACH: Ingestion of ground water

Pathway:	G Water	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	0.00E+00	0.00E+00
Fe-55	0.00E+00	0.00E+00
Co-60	0.00E+00	0.00E+00
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	0.00E+00	0.00E+00
Sr-90+D	1.52E+00	5.51E-07
Nb-94	6.55E-04	6.35E-10
Mo-93	0.00E+00	0.00E+00
Tc-99	0.00E+00	0.00E+00
Ru-106+D	0.00E+00	0.00E+00
Ag-110m	0.00E+00	0.00E+00
Sb-125	0.00E+00	0.00E+00
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	9.78E-01	1.37E-08
U-234	2.65E-02	4.15E-09
U-235+D	2.50E-02	4.39E-09
U-238+D	2.51E-02	5.78E-09
Np-237+D	4.16E-01	2.80E-08
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	9.87E-02	1.08E-08
U-Separ.	5.27E-02	1.01E-08
U-Deplete	2.79E-02	6.23E-09
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation SLAGROAD: Slag in road construction

Pathway:	External		Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mn-54	4.45E-02	3.39E-08	9.98E-07	5.50E-13	7.86E-06	5.57E-12
Fe-55	0.00E+00	0.00E+00	6.15E-09	1.28E-15	5.33E-08	3.09E-14
Co-60	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-59	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Zn-65	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sr-90+D	0.00E+00	0.00E+00	4.68E-05	2.49E-12	5.51E-04	2.00E-10
Nb-94	1.06E-01	8.06E-08	7.82E-05	1.55E-11	2.57E-05	2.49E-11
Mo-93	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Tc-99	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ru-106+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ag-110m	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sb-125	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	5.46E-03	4.15E-09	4.59E-07	2.87E-13	1.39E-05	8.97E-12
Cs-137+D	1.97E-03	1.50E-09	3.17E-07	1.90E-13	9.46E-06	5.99E-12
Ce-144+D	3.55E-03	2.70E-09	7.05E-05	2.03E-11	7.60E-05	1.07E-10
Pm-147	5.49E-07	4.18E-13	7.40E-06	1.41E-12	3.77E-06	5.09E-12
Eu-152	7.68E-02	5.84E-08	4.17E-05	1.49E-11	2.33E-05	2.06E-11
Pb-210+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-226+D	1.22E-01	9.31E-08	1.62E-03	5.18E-10	4.77E-03	1.06E-09
Ra-228+D	6.55E-02	4.98E-08	9.24E-04	1.81E-10	5.18E-03	8.90E-10
Ac-227+D	2.21E-02	1.68E-08	2.48E-01	2.90E-09	5.31E-02	2.26E-09
Th-228+D	1.12E-01	8.49E-08	6.51E-02	1.82E-08	2.91E-03	8.31E-10
Th-229+D	1.75E-02	1.33E-08	3.29E-01	1.25E-08	1.45E-02	1.29E-09
Th-230	1.32E-05	1.00E-11	4.94E-02	2.61E-09	1.97E-03	1.34E-10
Th-232	5.70E-06	4.33E-12	2.17E-01	2.55E-09	9.83E-03	1.18E-10
Pa-231	2.09E-03	1.59E-09	1.62E-01	3.05E-09	3.81E-02	5.35E-10
U-234	4.39E-06	3.34E-12	2.50E-02	2.63E-09	9.40E-05	1.47E-11
U-235+D	8.29E-03	6.31E-09	2.32E-02	2.45E-09	1.01E-04	1.77E-11
U-238+D	1.70E-03	1.30E-09	2.23E-02	2.35E-09	1.35E-04	3.10E-11
Np-237+D	1.20E-02	9.12E-09	1.02E-01	6.52E-09	1.60E-02	1.08E-09
Pu-238	1.66E-06	1.26E-12	5.44E-02	3.81E-09	1.78E-04	1.65E-11
Pu-239	3.23E-06	2.45E-12	5.82E-02	3.77E-09	1.86E-04	1.66E-11
Pu-240	1.61E-06	1.22E-12	5.82E-02	3.77E-09	1.86E-04	1.66E-11
Pu-241+D	2.08E-07	1.58E-13	9.36E-04	3.18E-11	2.76E-06	2.09E-13
Pu-242	1.40E-06	1.07E-12	5.53E-02	3.55E-09	1.77E-04	1.58E-11
Am-241	4.77E-04	3.63E-10	8.38E-02	7.26E-09	1.31E-02	1.18E-09
Cm-244	1.38E-06	1.05E-12	4.68E-02	4.59E-09	7.26E-03	7.58E-10
U-Series	1.26E-01	9.55E-08	1.19E-01	8.51E-09	1.13E-02	1.38E-09
U-Separ.	2.10E-03	1.60E-09	4.84E-02	5.10E-09	2.33E-04	4.66E-11
U-Deplete	1.84E-03	1.40E-09	2.50E-02	2.63E-09	1.45E-04	3.27E-11
Th-Series	1.77E-01	1.35E-07	2.83E-01	2.10E-08	1.79E-02	1.84E-09

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation ENGNWRKR: Manufacturing cars

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	1.80E-02	1.37E-08
Fe-55	0.00E+00	0.00E+00
Co-60	5.31E-02	4.04E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	2.49E-04	1.89E-10
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	2.33E-09	1.77E-15
Tc-99	2.41E-09	1.83E-15
Ru-106+D	4.61E-03	3.50E-09
Ag-110m	6.01E-02	4.57E-08
Sb-125	9.18E-03	6.98E-09
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation LATHEMFG: Manufacturing large industrial equipment

Pathway:	External		Inhalation		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	6.57E-07	2.20E-15	1.22E-05	6.02E-12
Mn-54	3.24E-02	2.47E-08	1.61E-06	8.85E-13	1.57E-05	1.11E-11
Fe-55	0.00E+00	0.00E+00	8.38E-07	1.74E-13	3.50E-06	2.03E-12
Co-60	1.06E-01	8.03E-08	1.04E-05	3.28E-12	1.57E-04	1.10E-10
Ni-59	0.00E+00	0.00E+00	8.52E-07	1.26E-13	1.22E-06	1.08E-12
Ni-63	0.00E+00	0.00E+00	1.98E-06	3.19E-13	3.37E-06	3.22E-12
Zn-65	4.77E-04	3.63E-10	1.27E-07	6.23E-14	1.67E-06	1.15E-12
Sr-90+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mo-93	8.22E-07	6.26E-13	3.17E-07	0.00E+00	7.86E-06	0.00E+00
Tc-99	2.43E-09	1.85E-15	3.23E-07	1.12E-13	8.53E-06	8.18E-12
Ru-106+D	7.80E-03	5.93E-09	1.77E-05	4.27E-12	1.60E-04	2.01E-10
Ag-110m	1.10E-01	8.38E-08	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Sb-125	1.48E-02	1.13E-08	6.70E-07	3.22E-13	1.63E-05	2.06E-11
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation COOKRNGE: End user of large home appliances

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	1.11E-02	8.41E-09
Fe-55	0.00E+00	0.00E+00
Co-60	6.32E-02	4.81E-08
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	1.99E-03	1.51E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	1.20E-07	9.16E-14
Tc-99	6.64E-09	5.05E-15
Ru-106+D	4.60E-03	3.50E-09
Ag-110m	4.97E-02	3.78E-08
Sb-125	2.15E-03	1.64E-09
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap Operation TAXIDRVR: End used of car

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	--	--
Mn-54	--	--
Fe-55	--	--
Co-60	5.43E-01	4.13E-07
Ni-59	--	--
Ni-63	--	--
Zn-65	--	--
Sr-90+D	--	--
Nb-94	--	--
Mo-93	--	--
Tc-99	--	--
Ru-106+D	3.66E-02	2.78E-08
Ag-110m	4.09E-01	3.11E-07
Sb-125	--	--
I-129	--	--
Cs-134	--	--
Cs-137+D	--	--
Ce-144+D	--	--
Pm-147	--	--
Eu-152	--	--
Pb-210+D	--	--
Ra-226+D	--	--
Ra-228+D	--	--
Ac-227+D	--	--
Th-228+D	--	--
Th-229+D	--	--
Th-230	--	--
Th-232	--	--
Pa-231	--	--
U-234	--	--
U-235+D	--	--
U-238+D	--	--
Np-237+D	--	--
Pu-238	--	--
Pu-239	--	--
Pu-240	--	--
Pu-241+D	--	--
Pu-242	--	--
Am-241	--	--
Cm-244	--	--
U-Series	--	--
U-Separ.	--	--
U-Deplete	--	--
Th-Series	--	--

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation OP-LATHE: End user of large industrial equipment

Pathway:	External	
Nuclide	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00
Mn-54	2.02E-01	1.54E-07
Fe-55	0.00E+00	0.00E+00
Co-60	8.99E-01	6.84E-07
Ni-59	0.00E+00	0.00E+00
Ni-63	0.00E+00	0.00E+00
Zn-65	2.70E-03	2.06E-09
Sr-90+D	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00
Mo-93	7.48E-06	5.69E-12
Tc-99	2.21E-08	1.68E-14
Ru-106+D	5.16E-02	3.93E-08
Ag-110m	6.29E-01	4.78E-07
Sb-125	2.23E-02	1.70E-08
I-129	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00

Individual Dose and Excess Cancer Morbidity per pCi/g of Scrap
Operation FEFRYPAN: End user of cast iron cooking utensils

Pathway:	External		Ingestion	
Nuclide	Dose (mrem/y)	Risk per year	Dose (mrem/y)	Risk per year
C-14	0.00E+00	0.00E+00	4.13E-06	2.04E-12
Mn-54	2.84E-04	2.16E-10	3.64E-06	2.58E-12
Fe-55	0.00E+00	0.00E+00	1.05E-06	6.07E-13
Co-60	1.12E-03	8.50E-10	4.99E-05	3.50E-11
Ni-59	0.00E+00	0.00E+00	4.12E-07	3.63E-13
Ni-63	0.00E+00	0.00E+00	1.14E-06	1.09E-12
Zn-65	3.51E-06	2.67E-12	3.52E-07	2.42E-13
Sr-90+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Nb-94	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Mo-93	4.12E-11	3.13E-17	2.66E-06	0.00E+00
Tc-99	6.03E-11	4.59E-17	2.89E-06	2.77E-12
Ru-106+D	7.85E-05	5.97E-11	3.94E-05	4.97E-11
Ag-110m	8.67E-04	6.59E-10	0.00E+00	0.00E+00
Sb-125	3.61E-05	2.75E-11	9.20E-07	1.16E-12
I-129	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-134	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cs-137+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ce-144+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pm-147	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Eu-152	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pb-210+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-226+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ra-228+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ac-227+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-228+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-229+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-230	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-232	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pa-231	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-234	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-235+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-238+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Np-237+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-238	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-239	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-240	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-241+D	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Pu-242	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Am-241	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Cm-244	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Series	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Separ.	0.00E+00	0.00E+00	0.00E+00	0.00E+00
U-Deplete	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Th-Series	0.00E+00	0.00E+00	0.00E+00	0.00E+00

APPENDIX K

MAXIMALLY EXPOSED INDIVIDUAL DOSES AND RISKS

Table of Contents

Table K-1: RME Individual Total Dose (mrem/y per pCi/g)

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLGLEACH	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
C-14	0.0e+00	1.5e-05	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	1.3e-05	0.0e+00	0.0e+00	0.0e+00	4.1e-06
Mn-54	5.8e-03	1.3e-01	6.9e-03	1.3e-04	1.3e-02	7.2e-03	9.5e-04	2.0e-01	0.0e+00	4.5e-02	1.8e-02	3.3e-02	1.1e-02	0.0e+00	2.0e-01	2.9e-04
Fe-55	0.0e+00	6.7e-06	2.5e-06	2.6e-06	2.6e-06	2.3e-06	0.0e+00	7.4e-07	0.0e+00	5.9e-08	0.0e+00	4.3e-06	0.0e+00	0.0e+00	0.0e+00	1.1e-06
Co-60	1.8e-02	4.2e-01	2.2e-02	7.4e-04	6.7e-02	2.2e-02	8.8e-04	0.0e+00	0.0e+00	0.0e+00	5.3e-02	1.1e-01	6.3e-02	5.4e-01	9.0e-01	1.2e-03
Ni-59	0.0e+00	4.4e-06	9.2e-07	9.9e-07	9.8e-07	9.3e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	2.1e-06	0.0e+00	0.0e+00	0.0e+00	4.1e-07
Ni-63	0.0e+00	1.1e-05	2.5e-06	2.7e-06	2.7e-06	2.5e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	5.4e-06	0.0e+00	0.0e+00	0.0e+00	1.1e-06
Zn-65	4.2e-03	9.6e-02	1.1e-02	5.8e-03	8.6e-03	8.0e-02	1.9e-02	0.0e+00	0.0e+00	0.0e+00	2.5e-04	4.8e-04	2.0e-03	0.0e+00	2.7e-03	3.9e-06
Sr-90+D	0.0e+00	2.4e-03	2.4e-03	2.5e-03	2.5e-03	2.2e-03	0.0e+00	8.7e-03	3.0e+00	6.0e-04	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Nb-94	1.1e-02	2.5e-01	1.3e-02	5.7e-04	4.0e-04	8.0e-03	2.1e-03	4.7e-01	1.3e-03	1.1e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mo-93	1.2e-07	5.7e-05	4.3e-06	6.6e-06	6.0e-06	8.3e-06	8.1e-09	0.0e+00	0.0e+00	0.0e+00	2.3e-09	9.0e-06	1.2e-07	0.0e+00	7.5e-06	2.7e-06
Tc-99	4.3e-10	2.2e-05	6.6e-06	7.3e-06	7.2e-06	7.0e-06	3.0e-11	0.0e+00	0.0e+00	0.0e+00	2.4e-09	8.9e-06	6.6e-09	0.0e+00	2.2e-08	2.9e-06
Ru-106+D	1.4e-03	3.4e-02	1.8e-03	2.1e-04	4.8e-03	1.8e-03	6.9e-05	0.0e+00	0.0e+00	0.0e+00	4.6e-03	8.0e-03	4.6e-03	3.7e-02	5.2e-02	1.2e-04
Ag-110m	1.9e-02	4.5e-01	2.4e-02	1.6e-03	6.9e-02	1.0e-01	2.3e-02	0.0e+00	0.0e+00	0.0e+00	6.0e-02	1.1e-01	5.0e-02	4.1e-01	6.3e-01	8.7e-04
Sb-125	2.6e-03	6.4e-02	3.2e-03	2.7e-04	8.9e-03	1.2e-02	2.6e-03	0.0e+00	0.0e+00	0.0e+00	9.2e-03	1.5e-02	2.2e-03	0.0e+00	2.2e-02	3.7e-05
I-129	1.9e-06	2.2e-03	1.7e-06	1.0e-28	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	1.1e-02	2.5e-01	4.0e-02	2.8e-02	2.8e-02	2.1e-01	5.0e-02	2.5e-02	0.0e+00	5.5e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-137+D	3.7e-03	8.9e-02	2.3e-02	1.9e-02	1.9e-02	8.4e-02	1.8e-02	8.9e-03	0.0e+00	2.0e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ce-144+D	2.6e-04	9.0e-03	7.9e-04	6.2e-04	5.8e-04	8.3e-04	5.1e-05	1.8e-02	0.0e+00	3.7e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pm-147	5.5e-09	5.3e-05	3.4e-05	4.6e-05	4.3e-05	5.3e-05	1.4e-09	1.4e-04	0.0e+00	1.2e-05	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Eu-152	7.5e-03	1.8e-01	8.7e-03	4.7e-04	2.5e-04	5.5e-03	1.5e-03	3.4e-01	0.0e+00	7.7e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pb-210+D	4.4e-07	7.0e-02	2.9e+00	3.1e+00	3.0e+00	2.8e+00	3.1e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	1.2e-02	3.1e-01	3.8e-02	2.7e-02	2.6e-02	3.4e-02	2.3e-03	6.3e-01	0.0e+00	1.3e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-1: RME Individual Total Dose (mrem/y per pCi/g)

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLGLEACH	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
Ra-228+D	6.2e-03	1.7e-01	3.1e-02	2.6e-02	2.5e-02	2.8e-02	1.2e-03	3.7e-01	0.0e+00	7.2e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ac-227+D	1.7e-03	8.0e+00	8.2e-01	1.2e+00	1.1e+00	1.5e+00	3.7e-04	3.9e+00	0.0e+00	3.2e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-228+D	1.0e-02	6.7e-01	1.8e-01	2.8e-01	2.5e-01	3.6e-01	2.0e-03	1.4e+00	0.0e+00	1.8e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-229+D	1.4e-03	2.6e+00	8.5e-01	1.4e+00	1.3e+00	1.8e+00	2.9e-04	4.4e+00	0.0e+00	3.6e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-230	4.4e-07	3.8e-01	1.3e-01	2.1e-01	1.9e-01	2.7e-01	1.1e-07	6.4e-01	0.0e+00	5.1e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-232	1.3e-07	1.9e+00	5.6e-01	9.2e-01	8.4e-01	1.2e+00	3.6e-08	2.8e+00	0.0e+00	2.3e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pa-231	1.3e-04	1.6e+00	5.5e-01	8.2e-01	7.6e-01	1.0e+00	2.8e-05	2.5e+00	2.0e+00	2.0e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-234	1.0e-07	1.6e-01	6.0e-02	1.0e-01	9.3e-02	1.3e-01	2.7e-08	3.1e-01	5.3e-02	2.5e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-235+D	4.4e-04	1.7e-01	5.7e-02	9.5e-02	8.6e-02	1.2e-01	1.1e-04	3.3e-01	5.0e-02	3.2e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-238+D	1.3e-04	1.4e-01	5.4e-02	9.1e-02	8.3e-02	1.2e-01	2.5e-05	2.9e-01	5.0e-02	2.4e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Np-237+D	9.4e-04	6.9e-01	3.1e-01	4.8e-01	4.4e-01	6.0e-01	2.0e-04	1.5e+00	8.3e-01	1.3e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-238	1.6e-08	4.8e-01	1.3e-01	2.2e-01	2.0e-01	2.9e-01	4.6e-09	6.8e-01	0.0e+00	5.5e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-239	7.8e-08	5.2e-01	1.4e-01	2.4e-01	2.2e-01	3.1e-01	2.1e-08	7.3e-01	0.0e+00	5.8e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-240	1.6e-08	5.2e-01	1.4e-01	2.4e-01	2.2e-01	3.1e-01	4.6e-09	7.3e-01	0.0e+00	5.8e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-241+D	6.8e-09	1.0e-02	2.3e-03	3.8e-03	3.5e-03	5.0e-03	1.6e-09	1.2e-02	0.0e+00	9.4e-04	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-242	1.5e-08	5.0e-01	1.3e-01	2.3e-01	2.1e-01	3.0e-01	4.2e-09	6.9e-01	0.0e+00	5.6e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Am-241	7.3e-06	5.4e-01	2.6e-01	4.0e-01	3.6e-01	4.9e-01	2.3e-06	1.2e+00	0.0e+00	9.7e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cm-244	1.4e-08	3.0e-01	1.4e-01	2.2e-01	2.0e-01	2.8e-01	3.8e-09	6.8e-01	0.0e+00	5.4e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Series	1.2e-02	1.5e+00	3.3e+00	3.6e+00	3.5e+00	3.5e+00	2.4e-03	2.2e+00	2.0e-01	2.6e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Separ.	1.5e-04	3.1e-01	1.2e-01	2.0e-01	1.8e-01	2.6e-01	3.0e-05	6.2e-01	1.1e-01	5.1e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Deplet	1.4e-04	1.6e-01	6.1e-02	1.0e-01	9.3e-02	1.3e-01	2.7e-05	3.2e-01	5.6e-02	2.7e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-Serie	1.7e-02	2.8e+00	7.7e-01	1.2e+00	1.1e+00	1.6e+00	3.2e-03	4.6e+00	0.0e+00	4.8e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-2: RME Individual Dose from External Exposure (mrem/y per pCi/g)

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
C-14	0.0e+00	3.5e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mn-54	5.8e-03	1.3e-01	6.9e-03	8.8e-05	1.3e-02	7.2e-03	9.5e-04	2.0e-01	4.5e-02	1.8e-02	3.2e-02	1.1e-02	0.0e+00	2.0e-01	2.8e-04
Fe-55	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Co-60	1.8e-02	4.2e-01	2.2e-02	6.5e-04	6.7e-02	2.2e-02	8.8e-04	0.0e+00	0.0e+00	5.3e-02	1.1e-01	6.3e-02	5.4e-01	9.0e-01	1.1e-03
Ni-59	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ni-63	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Zn-65	4.2e-03	9.6e-02	5.0e-03	1.2e-04	3.0e-03	7.5e-02	1.9e-02	0.0e+00	0.0e+00	2.5e-04	4.8e-04	2.0e-03	0.0e+00	2.7e-03	3.5e-06
Sr-90+D	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	1.2e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Nb-94	1.1e-02	2.5e-01	1.3e-02	1.5e-04	0.0e+00	7.5e-03	2.1e-03	4.7e-01	1.1e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mo-93	1.2e-07	1.5e-05	4.7e-08	4.7e-29	5.1e-09	1.3e-07	8.1e-09	0.0e+00	0.0e+00	2.3e-09	8.2e-07	1.2e-07	0.0e+00	7.5e-06	4.1e-11
Tc-99	4.3e-10	3.3e-06	4.4e-10	1.7e-21	1.3e-09	4.9e-10	3.0e-11	0.0e+00	0.0e+00	2.4e-09	2.4e-09	6.6e-09	0.0e+00	2.2e-08	6.0e-11
Ru-106+D	1.4e-03	3.3e-02	1.6e-03	1.4e-05	4.6e-03	1.6e-03	6.9e-05	0.0e+00	0.0e+00	4.6e-03	7.8e-03	4.6e-03	3.7e-02	5.2e-02	7.9e-05
Ag-110m	1.9e-02	4.5e-01	2.3e-02	4.0e-04	6.8e-02	1.0e-01	2.3e-02	0.0e+00	0.0e+00	6.0e-02	1.1e-01	5.0e-02	4.1e-01	6.3e-01	8.7e-04
Sb-125	2.6e-03	6.4e-02	3.0e-03	1.3e-05	8.6e-03	1.2e-02	2.6e-03	0.0e+00	0.0e+00	9.2e-03	1.5e-02	2.2e-03	0.0e+00	2.2e-02	3.6e-05
I-129	1.9e-06	3.4e-04	1.7e-06	1.0e-28	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	1.1e-02	2.5e-01	1.2e-02	1.2e-04	0.0e+00	1.9e-01	5.0e-02	2.4e-02	5.5e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-137+D	3.7e-03	8.9e-02	4.4e-03	3.2e-05	0.0e+00	6.8e-02	1.8e-02	8.8e-03	2.0e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ce-144+D	2.6e-04	8.4e-03	3.1e-04	1.3e-05	0.0e+00	1.8e-04	5.1e-05	1.6e-02	3.6e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pm-147	5.5e-09	1.3e-06	5.8e-09	3.7e-16	0.0e+00	4.9e-09	1.4e-09	2.5e-06	5.5e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Eu-152	7.5e-03	1.8e-01	8.6e-03	2.0e-04	0.0e+00	5.2e-03	1.5e-03	3.4e-01	7.7e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pb-210+D	4.4e-07	1.6e-04	4.2e-07	8.2e-10	0.0e+00	2.2e-05	3.1e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	1.2e-02	2.9e-01	1.4e-02	4.8e-04	0.0e+00	8.2e-03	2.3e-03	5.5e-01	1.2e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-2: RME Individual Dose from External Exposure (mrem/y per pCi/g)

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
Ra-228+D	6.2e-03	1.6e-01	7.1e-03	1.5e-04	0.0e+00	4.3e-03	1.2e-03	2.9e-01	6.6e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ac-227+D	1.7e-03	5.2e-02	2.0e-03	4.6e-06	0.0e+00	1.3e-03	3.7e-04	9.9e-02	2.2e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-228+D	1.0e-02	2.6e-01	1.2e-02	8.0e-04	0.0e+00	7.1e-03	2.0e-03	5.0e-01	1.1e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-229+D	1.4e-03	4.1e-02	1.4e-03	1.7e-05	0.0e+00	1.0e-03	2.9e-04	7.8e-02	1.8e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-230	4.4e-07	3.1e-05	4.6e-07	2.9e-12	0.0e+00	4.5e-07	1.1e-07	5.9e-05	1.3e-05	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-232	1.3e-07	1.4e-05	1.3e-07	1.1e-14	0.0e+00	1.6e-07	3.6e-08	2.5e-05	5.7e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pa-231	1.3e-04	5.0e-03	1.5e-04	7.3e-08	0.0e+00	9.8e-05	2.8e-05	9.3e-03	2.1e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-234	1.0e-07	1.0e-05	1.0e-07	5.4e-15	0.0e+00	1.2e-07	2.7e-08	2.0e-05	4.4e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-235+D	4.4e-04	2.0e-02	4.8e-04	1.1e-08	0.0e+00	3.6e-04	1.1e-04	3.7e-02	8.3e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-238+D	1.3e-04	4.0e-03	1.1e-04	2.5e-06	0.0e+00	8.9e-05	2.5e-05	7.6e-03	1.7e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Np-237+D	9.4e-04	2.8e-02	1.1e-03	6.8e-07	0.0e+00	7.1e-04	2.0e-04	5.4e-02	1.2e-02	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-238	1.6e-08	3.9e-06	9.4e-09	5.5e-30	0.0e+00	4.3e-08	4.6e-09	7.4e-06	1.7e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-239	7.8e-08	7.6e-06	7.9e-08	9.7e-16	0.0e+00	7.8e-08	2.1e-08	1.4e-05	3.2e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-240	1.6e-08	3.8e-06	9.7e-09	5.2e-30	0.0e+00	4.3e-08	4.6e-09	7.2e-06	1.6e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-241+D	6.8e-09	4.9e-07	7.4e-09	8.6e-13	0.0e+00	5.8e-09	1.6e-09	9.3e-07	2.1e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-242	1.5e-08	3.3e-06	9.0e-09	4.3e-30	0.0e+00	3.7e-08	4.2e-09	6.3e-06	1.4e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Am-241	7.3e-06	1.1e-03	7.0e-06	4.4e-27	0.0e+00	1.3e-05	2.3e-06	2.1e-03	4.8e-04	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cm-244	1.4e-08	3.3e-06	7.0e-09	5.4e-30	0.0e+00	3.8e-08	3.8e-09	6.2e-06	1.4e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Series	1.2e-02	3.0e-01	1.4e-02	4.8e-04	0.0e+00	8.4e-03	2.4e-03	5.6e-01	1.3e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Separ.	1.5e-04	5.0e-03	1.3e-04	2.5e-06	0.0e+00	1.1e-04	3.0e-05	9.3e-03	2.1e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Deplet	1.4e-04	4.3e-03	1.2e-04	2.5e-06	0.0e+00	9.4e-05	2.7e-05	8.2e-03	1.8e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-Serie	1.7e-02	4.2e-01	1.9e-02	9.5e-04	0.0e+00	1.1e-02	3.2e-03	7.9e-01	1.8e-01	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-3: RME Individual Dose from Inhalation (mrem/y per pCi/g)

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLAGROAD
C-14	2.4e-06	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mn-54	7.8e-06	2.6e-06	4.4e-06	4.0e-06	5.7e-06	1.3e-05	1.0e-06
Fe-55	3.1e-06	1.6e-07	2.6e-07	2.4e-07	3.5e-07	7.7e-08	6.2e-09
Co-60	2.6e-04	2.6e-05	4.3e-05	3.9e-05	5.7e-05	0.0e+00	0.0e+00
Ni-59	3.2e-06	1.1e-07	1.8e-07	1.7e-07	2.4e-07	0.0e+00	0.0e+00
Ni-63	7.4e-06	2.7e-07	4.5e-07	4.1e-07	6.0e-07	0.0e+00	0.0e+00
Zn-65	2.4e-05	2.3e-04	3.8e-04	3.5e-04	5.0e-04	0.0e+00	0.0e+00
Sr-90+D	1.5e-03	1.1e-04	1.9e-04	1.7e-04	2.5e-04	5.8e-04	4.7e-05
Nb-94	4.9e-04	1.9e-04	3.2e-04	2.9e-04	4.2e-04	9.8e-04	7.8e-05
Mo-93	3.3e-05	3.3e-06	5.6e-06	5.1e-06	7.4e-06	0.0e+00	0.0e+00
Tc-99	9.7e-06	9.7e-07	1.6e-06	1.5e-06	2.2e-06	0.0e+00	0.0e+00
Ru-106+D	5.6e-04	5.6e-05	9.4e-05	8.6e-05	1.2e-04	0.0e+00	0.0e+00
Ag-110m	9.4e-05	1.1e-04	1.9e-04	1.8e-04	2.5e-04	0.0e+00	0.0e+00
Sb-125	1.4e-05	2.8e-05	4.8e-05	4.3e-05	6.2e-05	0.0e+00	0.0e+00
I-129	2.0e-04	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	5.4e-05	5.1e-04	8.6e-04	7.8e-04	1.1e-03	5.7e-06	4.6e-07
Cs-137+D	3.7e-05	3.5e-04	6.0e-04	5.4e-04	7.8e-04	4.0e-06	3.2e-07
Ce-144+D	4.4e-04	1.7e-04	2.9e-04	2.6e-04	3.8e-04	8.8e-04	7.1e-05
Pm-147	4.6e-05	1.8e-05	3.0e-05	2.7e-05	3.9e-05	9.3e-05	7.4e-06
Eu-152	2.6e-04	1.0e-04	1.7e-04	1.5e-04	2.2e-04	5.2e-04	4.2e-05
Pb-210+D	2.7e-02	2.5e-01	4.2e-01	3.8e-01	5.5e-01	0.0e+00	0.0e+00
Ra-226+D	1.0e-02	3.9e-03	6.6e-03	6.0e-03	8.6e-03	2.0e-02	1.6e-03

Table K-3: RME Individual Dose from Inhalation (mrem/y per pCi/g)

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLAGROAD
Ra-228+D	5.9e-03	2.2e-03	3.8e-03	3.4e-03	4.9e-03	1.2e-02	9.2e-04
Ac-227+D	7.9e+00	6.0e-01	1.0e+00	9.2e-01	1.3e+00	3.1e+00	2.5e-01
Th-228+D	4.0e-01	1.6e-01	2.6e-01	2.4e-01	3.5e-01	8.1e-01	6.5e-02
Th-229+D	2.5e+00	7.9e-01	1.3e+00	1.2e+00	1.8e+00	4.1e+00	3.3e-01
Th-230	3.8e-01	1.2e-01	2.0e-01	1.8e-01	2.6e-01	6.2e-01	4.9e-02
Th-232	1.9e+00	5.2e-01	8.8e-01	8.0e-01	1.2e+00	2.7e+00	2.2e-01
Pa-231	1.5e+00	3.9e-01	6.6e-01	6.0e-01	8.6e-01	2.0e+00	1.6e-01
U-234	1.6e-01	6.0e-02	1.0e-01	9.2e-02	1.3e-01	3.1e-01	2.5e-02
U-235+D	1.4e-01	5.6e-02	9.4e-02	8.6e-02	1.2e-01	2.9e-01	2.3e-02
U-238+D	1.4e-01	5.4e-02	9.1e-02	8.3e-02	1.2e-01	2.8e-01	2.2e-02
Np-237+D	6.3e-01	2.5e-01	4.1e-01	3.8e-01	5.4e-01	1.3e+00	1.0e-01
Pu-238	4.6e-01	1.3e-01	2.2e-01	2.0e-01	2.9e-01	6.8e-01	5.4e-02
Pu-239	5.0e-01	1.4e-01	2.4e-01	2.2e-01	3.1e-01	7.3e-01	5.8e-02
Pu-240	5.0e-01	1.4e-01	2.4e-01	2.2e-01	3.1e-01	7.3e-01	5.8e-02
Pu-241+D	9.7e-03	2.3e-03	3.8e-03	3.5e-03	5.0e-03	1.2e-02	9.4e-04
Pu-242	4.8e-01	1.3e-01	2.3e-01	2.0e-01	2.9e-01	6.9e-01	5.5e-02
Am-241	5.2e-01	2.0e-01	3.4e-01	3.1e-01	4.5e-01	1.1e+00	8.4e-02
Cm-244	2.9e-01	1.1e-01	1.9e-01	1.7e-01	2.5e-01	5.9e-01	4.7e-02
U-Series	1.2e+00	5.3e-01	9.0e-01	8.2e-01	1.2e+00	1.5e+00	1.2e-01
U-Separ.	3.0e-01	1.2e-01	2.0e-01	1.8e-01	2.6e-01	6.1e-01	4.8e-02
U-Deplet	1.6e-01	6.0e-02	1.0e-01	9.2e-02	1.3e-01	3.1e-01	2.5e-02
Th-Serie	2.3e+00	6.8e-01	1.2e+00	1.1e+00	1.5e+00	3.5e+00	2.8e-01

Table K-4: RME Individual Dose from Ingestion (mrem/y per pCi/g)

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLGLEACH	SLAGROAD	LATHEMFG	FEFRYPAN
C-14	1.2e-05	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	1.2e-05	4.1e-06
Mn-54	1.6e-05	3.5e-05	3.5e-05	3.5e-05	3.0e-05	9.8e-05	0.0e+00	7.9e-06	1.6e-05	3.6e-06
Fe-55	3.6e-06	2.4e-06	2.4e-06	2.4e-06	2.0e-06	6.7e-07	0.0e+00	5.3e-08	3.5e-06	1.1e-06
Co-60	1.6e-04	4.0e-05	4.0e-05	4.0e-05	3.4e-05	0.0e+00	0.0e+00	0.0e+00	1.6e-04	5.0e-05
Ni-59	1.2e-06	8.1e-07	8.1e-07	8.1e-07	6.9e-07	0.0e+00	0.0e+00	0.0e+00	1.2e-06	4.1e-07
Ni-63	3.4e-06	2.2e-06	2.2e-06	2.2e-06	1.9e-06	0.0e+00	0.0e+00	0.0e+00	3.4e-06	1.1e-06
Zn-65	8.4e-05	5.3e-03	5.3e-03	5.3e-03	4.5e-03	0.0e+00	0.0e+00	0.0e+00	1.7e-06	3.5e-07
Sr-90+D	9.0e-04	2.3e-03	2.3e-03	2.3e-03	2.0e-03	6.9e-03	3.0e+00	5.5e-04	0.0e+00	0.0e+00
Nb-94	4.2e-05	1.1e-04	1.1e-04	1.1e-04	9.1e-05	3.2e-04	1.3e-03	2.6e-05	0.0e+00	0.0e+00
Mo-93	7.9e-06	9.3e-07	9.3e-07	9.3e-07	8.0e-07	0.0e+00	0.0e+00	0.0e+00	7.9e-06	2.7e-06
Tc-99	8.6e-06	5.7e-06	5.7e-06	5.7e-06	4.8e-06	0.0e+00	0.0e+00	0.0e+00	8.5e-06	2.9e-06
Ru-106+D	1.6e-04	1.1e-04	1.1e-04	1.1e-04	9.0e-05	0.0e+00	0.0e+00	0.0e+00	1.6e-04	3.9e-05
Ag-110m	6.3e-05	1.0e-03	1.0e-03	1.0e-03	8.8e-04	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Sb-125	1.6e-05	2.1e-04	2.1e-04	2.1e-04	1.8e-04	0.0e+00	0.0e+00	0.0e+00	1.6e-05	9.2e-07
I-129	1.6e-03	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	4.3e-04	2.7e-02	2.7e-02	2.7e-02	2.3e-02	1.7e-04	0.0e+00	1.4e-05	0.0e+00	0.0e+00
Cs-137+D	2.9e-04	1.8e-02	1.8e-02	1.8e-02	1.6e-02	1.2e-04	0.0e+00	9.5e-06	0.0e+00	0.0e+00
Ce-144+D	1.2e-04	3.2e-04	3.2e-04	3.2e-04	2.7e-04	9.5e-04	0.0e+00	7.6e-05	0.0e+00	0.0e+00
Pm-147	6.1e-06	1.6e-05	1.6e-05	1.6e-05	1.3e-05	4.7e-05	0.0e+00	3.8e-06	0.0e+00	0.0e+00
Eu-152	3.8e-05	9.7e-05	9.7e-05	9.7e-05	8.3e-05	2.9e-04	0.0e+00	2.3e-05	0.0e+00	0.0e+00
Pb-210+D	4.3e-02	2.7e+00	2.7e+00	2.7e+00	2.3e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	7.8e-03	2.0e-02	2.0e-02	2.0e-02	1.7e-02	6.0e-02	0.0e+00	4.8e-03	0.0e+00	0.0e+00

Table K-4: RME Individual Dose from Ingestion (mrem/y per pCi/g)

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLGLEACH	SLAGROAD	LATHEMFG	FEFRYPAN
Ra-228+D	8.4e-03	2.2e-02	2.2e-02	2.2e-02	1.8e-02	6.5e-02	0.0e+00	5.2e-03	0.0e+00	0.0e+00
Ac-227+D	8.6e-02	2.2e-01	2.2e-01	2.2e-01	1.9e-01	6.6e-01	0.0e+00	5.3e-02	0.0e+00	0.0e+00
Th-228+D	4.7e-03	1.2e-02	1.2e-02	1.2e-02	1.0e-02	3.6e-02	0.0e+00	2.9e-03	0.0e+00	0.0e+00
Th-229+D	2.4e-02	6.1e-02	6.1e-02	6.1e-02	5.2e-02	1.8e-01	0.0e+00	1.5e-02	0.0e+00	0.0e+00
Th-230	3.2e-03	8.2e-03	8.2e-03	8.2e-03	7.0e-03	2.5e-02	0.0e+00	2.0e-03	0.0e+00	0.0e+00
Th-232	1.6e-02	4.1e-02	4.1e-02	4.1e-02	3.5e-02	1.2e-01	0.0e+00	9.8e-03	0.0e+00	0.0e+00
Pa-231	6.2e-02	1.6e-01	1.6e-01	1.6e-01	1.4e-01	4.8e-01	2.0e+00	3.8e-02	0.0e+00	0.0e+00
U-234	1.7e-03	3.9e-04	3.9e-04	3.9e-04	3.3e-04	1.2e-03	5.3e-02	9.4e-05	0.0e+00	0.0e+00
U-235+D	1.6e-03	4.2e-04	4.2e-04	4.2e-04	3.6e-04	1.3e-03	5.0e-02	1.0e-04	0.0e+00	0.0e+00
U-238+D	1.6e-03	5.6e-04	5.6e-04	5.6e-04	4.8e-04	1.7e-03	5.0e-02	1.4e-04	0.0e+00	0.0e+00
Np-237+D	2.6e-02	6.7e-02	6.7e-02	6.7e-02	5.7e-02	2.0e-01	8.3e-01	1.6e-02	0.0e+00	0.0e+00
Pu-238	1.9e-02	7.4e-04	7.4e-04	7.4e-04	6.3e-04	2.2e-03	0.0e+00	1.8e-04	0.0e+00	0.0e+00
Pu-239	2.1e-02	7.8e-04	7.8e-04	7.8e-04	6.6e-04	2.3e-03	0.0e+00	1.9e-04	0.0e+00	0.0e+00
Pu-240	2.1e-02	7.8e-04	7.8e-04	7.8e-04	6.6e-04	2.3e-03	0.0e+00	1.9e-04	0.0e+00	0.0e+00
Pu-241+D	4.0e-04	1.2e-05	1.2e-05	1.2e-05	9.8e-06	3.5e-05	0.0e+00	2.8e-06	0.0e+00	0.0e+00
Pu-242	2.0e-02	7.4e-04	7.4e-04	7.4e-04	6.3e-04	2.2e-03	0.0e+00	1.8e-04	0.0e+00	0.0e+00
Am-241	2.1e-02	5.5e-02	5.5e-02	5.5e-02	4.7e-02	1.6e-01	0.0e+00	1.3e-02	0.0e+00	0.0e+00
Cm-244	1.2e-02	3.0e-02	3.0e-02	3.0e-02	2.6e-02	9.1e-02	0.0e+00	7.3e-03	0.0e+00	0.0e+00
U-Series	6.4e-02	2.7e+00	2.7e+00	2.7e+00	2.3e+00	1.4e-01	2.0e-01	1.1e-02	0.0e+00	0.0e+00
U-Separ.	3.3e-03	9.7e-04	9.7e-04	9.7e-04	8.3e-04	2.9e-03	1.1e-01	2.3e-04	0.0e+00	0.0e+00
U-Deplet	1.8e-03	6.0e-04	6.0e-04	6.0e-04	5.1e-04	1.8e-03	5.6e-02	1.5e-04	0.0e+00	0.0e+00
Th-Serie	2.9e-02	7.5e-02	7.5e-02	7.5e-02	6.4e-02	2.2e-01	0.0e+00	1.8e-02	0.0e+00	0.0e+00

Table K-5: RME Individual Total Risk

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLGLEACH	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
C-14	0.0e+00	6.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	6.0e-12	0.0e+00	0.0e+00	0.0e+00	2.0e-12
Mn-54	4.4e-09	1.0e-07	5.2e-09	9.5e-11	1.0e-08	5.5e-09	7.2e-10	1.5e-07	0.0e+00	3.4e-08	1.4e-08	2.5e-08	8.4e-09	0.0e+00	1.5e-07	2.2e-10
Fe-55	0.0e+00	2.7e-12	1.4e-12	1.4e-12	1.4e-12	1.2e-12	0.0e+00	4.0e-13	0.0e+00	3.2e-14	0.0e+00	2.2e-12	0.0e+00	0.0e+00	0.0e+00	6.1e-13
Co-60	1.4e-08	3.2e-07	1.7e-08	5.4e-10	5.1e-08	1.6e-08	6.7e-10	0.0e+00	0.0e+00	0.0e+00	4.0e-08	8.0e-08	4.8e-08	4.1e-07	6.8e-07	8.9e-10
Ni-59	0.0e+00	1.6e-12	7.3e-13	7.4e-13	7.4e-13	6.5e-13	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	1.2e-12	0.0e+00	0.0e+00	0.0e+00	3.6e-13
Ni-63	0.0e+00	4.4e-12	2.2e-12	2.2e-12	2.2e-12	1.9e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	3.5e-12	0.0e+00	0.0e+00	0.0e+00	1.1e-12
Zn-65	3.2e-09	7.3e-08	7.5e-09	3.9e-09	6.1e-09	6.0e-08	1.5e-08	0.0e+00	0.0e+00	0.0e+00	1.9e-10	3.6e-10	1.5e-09	0.0e+00	2.1e-09	2.9e-12
Sr-90+D	0.0e+00	4.1e-10	8.4e-10	8.5e-10	8.5e-10	7.3e-10	0.0e+00	3.5e-09	1.1e-06	2.0e-10	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Nb-94	8.2e-09	1.9e-07	9.9e-09	2.8e-10	1.6e-10	5.8e-09	1.6e-09	3.6e-07	1.3e-09	8.1e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mo-93	8.9e-14	1.2e-11	3.6e-14	0.0e+00	3.9e-15	9.8e-14	6.2e-15	0.0e+00	0.0e+00	0.0e+00	1.8e-15	6.3e-13	9.2e-14	0.0e+00	5.7e-12	3.1e-17
Tc-99	3.3e-16	1.4e-11	5.8e-12	6.0e-12	5.9e-12	5.4e-12	2.3e-17	0.0e+00	0.0e+00	0.0e+00	1.8e-15	8.3e-12	5.1e-15	0.0e+00	1.7e-14	2.8e-12
Ru-106+D	1.0e-09	2.6e-08	1.4e-09	1.7e-10	3.7e-09	1.4e-09	5.2e-11	0.0e+00	0.0e+00	0.0e+00	3.5e-09	6.1e-09	3.5e-09	2.8e-08	3.9e-08	1.1e-10
Ag-110m	1.5e-08	3.4e-07	1.8e-08	1.2e-09	5.2e-08	7.9e-08	1.8e-08	0.0e+00	0.0e+00	0.0e+00	4.6e-08	8.4e-08	3.8e-08	3.1e-07	4.8e-07	6.6e-10
Sb-125	2.0e-09	4.9e-08	2.6e-09	3.0e-10	6.8e-09	9.2e-09	2.0e-09	0.0e+00	0.0e+00	0.0e+00	7.0e-09	1.1e-08	1.6e-09	0.0e+00	1.7e-08	2.9e-11
I-129	1.5e-12	1.5e-09	1.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	8.0e-09	1.9e-07	2.7e-08	1.8e-08	1.8e-08	1.6e-07	3.8e-08	1.9e-08	0.0e+00	4.2e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-137+D	2.9e-09	6.8e-08	1.5e-08	1.2e-08	1.2e-08	6.2e-08	1.4e-08	6.8e-09	0.0e+00	1.5e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ce-144+D	1.9e-10	6.7e-09	7.3e-10	5.4e-10	5.2e-10	6.2e-10	3.9e-11	1.4e-08	0.0e+00	2.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pm-147	4.2e-15	1.8e-11	2.5e-11	2.7e-11	2.6e-11	2.6e-11	1.1e-15	8.3e-11	0.0e+00	6.9e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Eu-152	5.7e-09	1.4e-07	6.6e-09	3.0e-10	1.4e-10	4.1e-09	1.1e-09	2.6e-07	0.0e+00	5.8e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pb-210+D	3.3e-13	1.1e-08	4.1e-07	4.4e-07	4.3e-07	4.0e-07	2.4e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	9.1e-09	2.3e-07	1.6e-08	6.9e-09	6.4e-09	1.3e-08	1.8e-09	4.4e-07	0.0e+00	9.5e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-5: RME Individual Total Risk

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLGLEACH	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
Ra-228+D	4.8e-09	1.2e-07	9.5e-09	4.6e-09	4.4e-09	7.4e-09	9.2e-10	2.4e-07	0.0e+00	5.1e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ac-227+D	1.3e-09	1.4e-07	1.8e-08	2.1e-08	2.0e-08	2.4e-08	2.8e-10	1.4e-07	0.0e+00	2.2e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-228+D	7.9e-09	3.2e-07	5.6e-08	7.8e-08	7.1e-08	1.1e-07	1.5e-09	6.2e-07	0.0e+00	1.0e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-229+D	1.1e-09	1.3e-07	3.7e-08	5.6e-08	5.2e-08	7.2e-08	2.2e-10	2.3e-07	0.0e+00	2.7e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-230	3.3e-13	2.0e-08	6.8e-09	1.1e-08	1.0e-08	1.4e-08	8.7e-14	3.4e-08	0.0e+00	2.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-232	1.0e-13	2.3e-08	6.6e-09	1.1e-08	9.9e-09	1.4e-08	2.7e-14	3.3e-08	0.0e+00	2.7e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pa-231	9.9e-11	3.3e-08	9.7e-09	1.5e-08	1.4e-08	1.8e-08	2.1e-11	5.2e-08	2.8e-08	5.2e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-234	7.7e-14	1.7e-08	6.4e-09	1.1e-08	9.8e-09	1.4e-08	2.1e-14	3.3e-08	8.3e-09	2.7e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-235+D	3.4e-10	3.0e-08	6.3e-09	1.0e-08	9.1e-09	1.3e-08	8.1e-11	5.9e-08	8.8e-09	8.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-238+D	9.8e-11	1.8e-08	5.9e-09	9.7e-09	8.8e-09	1.3e-08	1.9e-11	3.6e-08	1.2e-08	3.7e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Np-237+D	7.1e-10	6.4e-08	2.1e-08	3.1e-08	2.9e-08	3.9e-08	1.6e-10	1.4e-07	5.6e-08	1.7e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-238	1.3e-14	3.4e-08	9.2e-09	1.6e-08	1.4e-08	2.0e-08	3.5e-15	4.8e-08	0.0e+00	3.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-239	5.9e-14	3.4e-08	9.1e-09	1.5e-08	1.4e-08	2.0e-08	1.6e-14	4.7e-08	0.0e+00	3.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-240	1.2e-14	3.4e-08	9.1e-09	1.5e-08	1.4e-08	2.0e-08	3.5e-15	4.7e-08	0.0e+00	3.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-241+D	5.2e-15	3.6e-10	7.7e-11	1.3e-10	1.2e-10	1.7e-10	1.3e-15	4.0e-10	0.0e+00	3.2e-11	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-242	1.1e-14	3.3e-08	8.6e-09	1.5e-08	1.3e-08	1.9e-08	3.2e-15	4.5e-08	0.0e+00	3.6e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Am-241	5.5e-12	4.8e-08	2.2e-08	3.4e-08	3.2e-08	4.3e-08	1.7e-12	1.1e-07	0.0e+00	8.8e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cm-244	1.1e-14	3.0e-08	1.4e-08	2.2e-08	2.0e-08	2.7e-08	2.9e-15	6.7e-08	0.0e+00	5.4e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Series	9.3e-09	3.0e-07	4.5e-07	4.8e-07	4.7e-07	4.6e-07	1.8e-09	5.5e-07	2.2e-08	1.1e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Separ.	1.1e-10	3.6e-08	1.3e-08	2.1e-08	1.9e-08	2.7e-08	2.3e-11	7.1e-08	2.0e-08	6.7e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Deplet	1.0e-10	2.0e-08	6.5e-09	1.1e-08	9.8e-09	1.4e-08	2.0e-11	4.0e-08	1.3e-08	4.1e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-Serie	1.3e-08	4.6e-07	7.3e-08	9.4e-08	8.5e-08	1.3e-07	2.4e-09	8.9e-07	0.0e+00	1.6e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-6: RME Individual Risk from External Exposure

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
C-14	0.0e+00	2.7e-13	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mn-54	4.4e-09	1.0e-07	5.2e-09	6.7e-11	1.0e-08	5.5e-09	7.2e-10	1.5e-07	3.4e-08	1.4e-08	2.5e-08	8.4e-09	0.0e+00	1.5e-07	2.2e-10
Fe-55	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Co-60	1.4e-08	3.2e-07	1.7e-08	5.0e-10	5.1e-08	1.6e-08	6.7e-10	0.0e+00	0.0e+00	4.0e-08	8.0e-08	4.8e-08	4.1e-07	6.8e-07	8.5e-10
Ni-59	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ni-63	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Zn-65	3.2e-09	7.3e-08	3.8e-09	8.8e-11	2.3e-09	5.7e-08	1.5e-08	0.0e+00	0.0e+00	1.9e-10	3.6e-10	1.5e-09	0.0e+00	2.1e-09	2.7e-12
Sr-90+D	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	9.1e-10	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Nb-94	8.2e-09	1.9e-07	9.7e-09	1.1e-10	0.0e+00	5.7e-09	1.6e-09	3.6e-07	8.1e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mo-93	8.9e-14	1.2e-11	3.6e-14	0.0e+00	3.9e-15	9.8e-14	6.2e-15	0.0e+00	0.0e+00	1.8e-15	6.3e-13	9.2e-14	0.0e+00	5.7e-12	3.1e-17
Tc-99	3.3e-16	2.5e-12	3.4e-16	1.3e-27	9.5e-16	3.7e-16	2.3e-17	0.0e+00	0.0e+00	1.8e-15	1.9e-15	5.1e-15	0.0e+00	1.7e-14	4.6e-17
Ru-106+D	1.0e-09	2.5e-08	1.2e-09	1.1e-11	3.5e-09	1.2e-09	5.2e-11	0.0e+00	0.0e+00	3.5e-09	5.9e-09	3.5e-09	2.8e-08	3.9e-08	6.0e-11
Ag-110m	1.5e-08	3.4e-07	1.7e-08	3.1e-10	5.2e-08	7.8e-08	1.8e-08	0.0e+00	0.0e+00	4.6e-08	8.4e-08	3.8e-08	3.1e-07	4.8e-07	6.6e-10
Sb-125	2.0e-09	4.9e-08	2.3e-09	1.0e-11	6.5e-09	9.0e-09	2.0e-09	0.0e+00	0.0e+00	7.0e-09	1.1e-08	1.6e-09	0.0e+00	1.7e-08	2.8e-11
I-129	1.5e-12	2.6e-10	1.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	8.0e-09	1.9e-07	9.4e-09	9.2e-11	0.0e+00	1.4e-07	3.8e-08	1.9e-08	4.2e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-137+D	2.9e-09	6.8e-08	3.3e-09	2.5e-11	0.0e+00	5.2e-08	1.4e-08	6.7e-09	1.5e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ce-144+D	1.9e-10	6.4e-09	2.3e-10	1.0e-11	0.0e+00	1.4e-10	3.9e-11	1.2e-08	2.7e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pm-147	4.2e-15	9.9e-13	4.4e-15	2.8e-22	0.0e+00	3.7e-15	1.1e-15	1.9e-12	4.2e-13	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Eu-152	5.7e-09	1.4e-07	6.5e-09	1.6e-10	0.0e+00	4.0e-09	1.1e-09	2.6e-07	5.8e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pb-210+D	3.3e-13	1.2e-10	3.2e-13	6.2e-16	0.0e+00	1.7e-11	2.4e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	9.1e-09	2.2e-07	1.1e-08	3.7e-10	0.0e+00	6.2e-09	1.8e-09	4.2e-07	9.3e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-6: RME Individual Risk from External Exposure

	SCRDRIVE	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	DUSTDRIV	SLAGPILE	SLAGROAD	ENGNWRKR	LATHEMFG	COOKRNGE	TAXIDRVR	OP-LATHE	FEFRYPAN
Ra-228+D	4.8e-09	1.2e-07	5.4e-09	1.1e-10	0.0e+00	3.3e-09	9.2e-10	2.2e-07	5.0e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ac-227+D	1.3e-09	4.0e-08	1.6e-09	3.5e-12	0.0e+00	9.9e-10	2.8e-10	7.5e-08	1.7e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-228+D	7.9e-09	2.0e-07	9.1e-09	6.1e-10	0.0e+00	5.4e-09	1.5e-09	3.8e-07	8.5e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-229+D	1.1e-09	3.2e-08	1.0e-09	1.3e-11	0.0e+00	7.8e-10	2.2e-10	5.9e-08	1.3e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-230	3.3e-13	2.4e-11	3.5e-13	2.2e-18	0.0e+00	3.4e-13	8.7e-14	4.5e-11	1.0e-11	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-232	1.0e-13	1.0e-11	1.0e-13	8.3e-21	0.0e+00	1.2e-13	2.7e-14	1.9e-11	4.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pa-231	9.9e-11	3.8e-09	1.1e-10	5.6e-14	0.0e+00	7.4e-11	2.1e-11	7.1e-09	1.6e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-234	7.7e-14	7.9e-12	7.6e-14	4.1e-21	0.0e+00	9.4e-14	2.1e-14	1.5e-11	3.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-235+D	3.4e-10	1.5e-08	3.7e-10	8.0e-15	0.0e+00	2.7e-10	8.1e-11	2.8e-08	6.3e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-238+D	9.8e-11	3.1e-09	8.4e-11	1.9e-12	0.0e+00	6.7e-11	1.9e-11	5.8e-09	1.3e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Np-237+D	7.1e-10	2.2e-08	8.0e-10	5.2e-13	0.0e+00	5.4e-10	1.6e-10	4.1e-08	9.1e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-238	1.3e-14	3.0e-12	7.2e-15	0.0e+00	0.0e+00	3.3e-14	3.5e-15	5.6e-12	1.3e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-239	5.9e-14	5.8e-12	6.0e-14	7.4e-22	0.0e+00	5.9e-14	1.6e-14	1.1e-11	2.5e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-240	1.2e-14	2.9e-12	7.4e-15	0.0e+00	0.0e+00	3.3e-14	3.5e-15	5.5e-12	1.2e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-241+D	5.2e-15	3.7e-13	5.7e-15	6.6e-19	0.0e+00	4.4e-15	1.3e-15	7.0e-13	1.6e-13	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Pu-242	1.1e-14	2.5e-12	6.8e-15	0.0e+00	0.0e+00	2.8e-14	3.2e-15	4.8e-12	1.1e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Am-241	5.5e-12	8.6e-10	5.3e-12	0.0e+00	0.0e+00	9.7e-12	1.7e-12	1.6e-09	3.6e-10	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cm-244	1.1e-14	2.5e-12	5.3e-15	0.0e+00	0.0e+00	2.9e-14	2.9e-15	4.7e-12	1.1e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Series	9.3e-09	2.3e-07	1.1e-08	3.7e-10	0.0e+00	6.4e-09	1.8e-09	4.3e-07	9.6e-08	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Separ.	1.1e-10	3.8e-09	1.0e-10	1.9e-12	0.0e+00	8.0e-11	2.3e-11	7.1e-09	1.6e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
U-Deplet	1.0e-10	3.3e-09	9.0e-11	1.9e-12	0.0e+00	7.2e-11	2.0e-11	6.2e-09	1.4e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Th-Serie	1.3e-08	3.2e-07	1.5e-08	7.2e-10	0.0e+00	8.7e-09	2.4e-09	6.0e-07	1.4e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00

Table K-8: RME Individual Risk Ingestion

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLGLEACH	SLAGROAD	LATHEMFG	FEFRYPAN
C-14	6.0e-12	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	6.0e-12	2.0e-12
Mn-54	1.2e-11	2.5e-11	2.5e-11	2.5e-11	2.1e-11	7.0e-11	0.0e+00	5.6e-12	1.1e-11	2.6e-12
Fe-55	2.1e-12	1.4e-12	1.4e-12	1.4e-12	1.2e-12	3.9e-13	0.0e+00	3.1e-14	2.0e-12	6.1e-13
Co-60	1.1e-10	2.8e-11	2.8e-11	2.8e-11	2.4e-11	0.0e+00	0.0e+00	0.0e+00	1.1e-10	3.5e-11
Ni-59	1.1e-12	7.2e-13	7.2e-13	7.2e-13	6.1e-13	0.0e+00	0.0e+00	0.0e+00	1.1e-12	3.6e-13
Ni-63	3.2e-12	2.1e-12	2.1e-12	2.1e-12	1.8e-12	0.0e+00	0.0e+00	0.0e+00	3.2e-12	1.1e-12
Zn-65	5.8e-11	3.6e-09	3.6e-09	3.6e-09	3.1e-09	0.0e+00	0.0e+00	0.0e+00	1.2e-12	2.4e-13
Sr-90+D	3.3e-10	8.4e-10	8.4e-10	8.4e-10	7.1e-10	2.5e-09	1.1e-06	2.0e-10	0.0e+00	0.0e+00
Nb-94	4.1e-11	1.0e-10	1.0e-10	1.0e-10	8.9e-11	3.1e-10	1.3e-09	2.5e-11	0.0e+00	0.0e+00
Mo-93	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Tc-99	8.2e-12	5.4e-12	5.4e-12	5.4e-12	4.6e-12	0.0e+00	0.0e+00	0.0e+00	8.2e-12	2.8e-12
Ru-106+D	2.0e-10	1.3e-10	1.3e-10	1.3e-10	1.1e-10	0.0e+00	0.0e+00	0.0e+00	2.0e-10	5.0e-11
Ag-110m	4.9e-11	8.0e-10	8.0e-10	8.0e-10	6.8e-10	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Sb-125	2.1e-11	2.7e-10	2.7e-10	2.7e-10	2.3e-10	0.0e+00	0.0e+00	0.0e+00	2.1e-11	1.2e-12
I-129	1.1e-09	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	2.8e-10	1.7e-08	1.7e-08	1.7e-08	1.5e-08	1.1e-10	0.0e+00	9.0e-12	0.0e+00	0.0e+00
Cs-137+D	1.9e-10	1.2e-08	1.2e-08	1.2e-08	9.8e-09	7.5e-11	0.0e+00	6.0e-12	0.0e+00	0.0e+00
Ce-144+D	1.7e-10	4.5e-10	4.5e-10	4.5e-10	3.8e-10	1.3e-09	0.0e+00	1.1e-10	0.0e+00	0.0e+00
Pm-147	8.3e-12	2.1e-11	2.1e-11	2.1e-11	1.8e-11	6.4e-11	0.0e+00	5.1e-12	0.0e+00	0.0e+00
Eu-152	3.4e-11	8.6e-11	8.6e-11	8.6e-11	7.3e-11	2.6e-10	0.0e+00	2.1e-11	0.0e+00	0.0e+00
Pb-210+D	5.9e-09	3.7e-07	3.7e-07	3.7e-07	3.1e-07	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Ra-226+D	1.7e-09	4.4e-09	4.4e-09	4.4e-09	3.8e-09	1.3e-08	0.0e+00	1.1e-09	0.0e+00	0.0e+00

Table k-8: RME Individual Risk Ingestion

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLGLEACH	SLAGROAD	LATHEMFG	FEFRYPAN
Ra-228+D	1.5e-09	3.7e-09	3.7e-09	3.7e-09	3.2e-09	1.1e-08	0.0e+00	8.9e-10	0.0e+00	0.0e+00
Ac-227+D	3.7e-09	9.4e-09	9.4e-09	9.4e-09	8.0e-09	2.8e-08	0.0e+00	2.3e-09	0.0e+00	0.0e+00
Th-228+D	1.4e-09	3.5e-09	3.5e-09	3.5e-09	3.0e-09	1.0e-08	0.0e+00	8.3e-10	0.0e+00	0.0e+00
Th-229+D	2.1e-09	5.4e-09	5.4e-09	5.4e-09	4.6e-09	1.6e-08	0.0e+00	1.3e-09	0.0e+00	0.0e+00
Th-230	2.2e-10	5.6e-10	5.6e-10	5.6e-10	4.8e-10	1.7e-09	0.0e+00	1.3e-10	0.0e+00	0.0e+00
Th-232	1.9e-10	4.9e-10	4.9e-10	4.9e-10	4.2e-10	1.5e-09	0.0e+00	1.2e-10	0.0e+00	0.0e+00
Pa-231	8.7e-10	2.2e-09	2.2e-09	2.2e-09	1.9e-09	6.7e-09	2.8e-08	5.4e-10	0.0e+00	0.0e+00
U-234	2.6e-10	6.1e-11	6.1e-11	6.1e-11	5.2e-11	1.8e-10	8.3e-09	1.5e-11	0.0e+00	0.0e+00
U-235+D	2.8e-10	7.4e-11	7.4e-11	7.4e-11	6.3e-11	2.2e-10	8.8e-09	1.8e-11	0.0e+00	0.0e+00
U-238+D	3.6e-10	1.3e-10	1.3e-10	1.3e-10	1.1e-10	3.9e-10	1.2e-08	3.1e-11	0.0e+00	0.0e+00
Np-237+D	1.8e-09	4.5e-09	4.5e-09	4.5e-09	3.8e-09	1.4e-08	5.6e-08	1.1e-09	0.0e+00	0.0e+00
Pu-238	1.7e-09	6.9e-11	6.9e-11	6.9e-11	5.9e-11	2.1e-10	0.0e+00	1.7e-11	0.0e+00	0.0e+00
Pu-239	1.9e-09	6.9e-11	6.9e-11	6.9e-11	5.9e-11	2.1e-10	0.0e+00	1.7e-11	0.0e+00	0.0e+00
Pu-240	1.8e-09	6.9e-11	6.9e-11	6.9e-11	5.9e-11	2.1e-10	0.0e+00	1.7e-11	0.0e+00	0.0e+00
Pu-241+D	3.0e-11	8.7e-13	8.7e-13	8.7e-13	7.4e-13	2.6e-12	0.0e+00	2.1e-13	0.0e+00	0.0e+00
Pu-242	1.8e-09	6.6e-11	6.6e-11	6.6e-11	5.6e-11	2.0e-10	0.0e+00	1.6e-11	0.0e+00	0.0e+00
Am-241	1.9e-09	4.9e-09	4.9e-09	4.9e-09	4.2e-09	1.5e-08	0.0e+00	1.2e-09	0.0e+00	0.0e+00
Cm-244	1.2e-09	3.2e-09	3.2e-09	3.2e-09	2.7e-09	9.5e-09	0.0e+00	7.6e-10	0.0e+00	0.0e+00
U-Series	8.7e-09	3.7e-07	3.7e-07	3.7e-07	3.2e-07	1.7e-08	2.2e-08	1.4e-09	0.0e+00	0.0e+00
U-Separ.	6.3e-10	1.9e-10	1.9e-10	1.9e-10	1.7e-10	5.8e-10	2.0e-08	4.7e-11	0.0e+00	0.0e+00
U-Deplet	3.9e-10	1.4e-10	1.4e-10	1.4e-10	1.2e-10	4.1e-10	1.3e-08	3.3e-11	0.0e+00	0.0e+00
Th-Serie	3.0e-09	7.7e-09	7.7e-09	7.7e-09	6.5e-09	2.3e-08	0.0e+00	1.8e-09	0.0e+00	0.0e+00

Table K-7: RME Individual Risk from Inhalation

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLAGROAD
C-14	8.2e-15	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Mn-54	4.3e-12	1.4e-12	2.4e-12	2.2e-12	3.2e-12	6.9e-12	5.5e-13
Fe-55	6.5e-13	3.2e-14	5.5e-14	5.0e-14	7.2e-14	1.6e-14	1.3e-15
Co-60	8.1e-11	8.0e-12	1.4e-11	1.2e-11	1.8e-11	0.0e+00	0.0e+00
Ni-59	4.7e-13	1.6e-14	2.7e-14	2.4e-14	3.5e-14	0.0e+00	0.0e+00
Ni-63	1.2e-12	4.3e-14	7.3e-14	6.7e-14	9.6e-14	0.0e+00	0.0e+00
Zn-65	1.2e-11	1.1e-10	1.9e-10	1.7e-10	2.4e-10	0.0e+00	0.0e+00
Sr-90+D	8.1e-11	6.0e-12	1.0e-11	9.2e-12	1.3e-11	3.1e-11	2.5e-12
Nb-94	9.6e-11	3.7e-11	6.3e-11	5.7e-11	8.3e-11	1.9e-10	1.6e-11
Mo-93	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Tc-99	3.4e-12	3.4e-13	5.7e-13	5.2e-13	7.5e-13	0.0e+00	0.0e+00
Ru-106+D	1.4e-10	1.3e-11	2.3e-11	2.1e-11	3.0e-11	0.0e+00	0.0e+00
Ag-110m	3.8e-11	4.6e-11	7.7e-11	7.0e-11	1.0e-10	0.0e+00	0.0e+00
Sb-125	6.9e-12	1.4e-11	2.3e-11	2.1e-11	3.0e-11	0.0e+00	0.0e+00
I-129	1.4e-10	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00	0.0e+00
Cs-134	3.4e-11	3.2e-10	5.4e-10	4.9e-10	7.1e-10	3.6e-12	2.9e-13
Cs-137+D	2.2e-11	2.1e-10	3.6e-10	3.3e-10	4.7e-10	2.4e-12	1.9e-13
Ce-144+D	1.3e-10	4.9e-11	8.3e-11	7.5e-11	1.1e-10	2.5e-10	2.0e-11
Pm-147	8.7e-12	3.4e-12	5.7e-12	5.2e-12	7.5e-12	1.8e-11	1.4e-12
Eu-152	9.3e-11	3.6e-11	6.1e-11	5.5e-11	8.0e-11	1.9e-10	1.5e-11
Pb-210+D	4.5e-09	4.1e-08	7.0e-08	6.3e-08	9.1e-08	0.0e+00	0.0e+00
Ra-226+D	3.2e-09	1.2e-09	2.1e-09	1.9e-09	2.8e-09	6.5e-09	5.2e-10

Table K-7: RME Individual Risk from Inhalation

	SCRAPCUT	OP-CRANE	FURNACE	OPCASTER	BAGHOUSE	SLAGPILE	SLAGROAD
Ra-228+D	1.2e-09	4.3e-10	7.4e-10	6.7e-10	9.6e-10	2.3e-09	1.8e-10
Ac-227+D	9.2e-08	7.0e-09	1.2e-08	1.1e-08	1.5e-08	3.6e-08	2.9e-09
Th-228+D	1.1e-07	4.4e-08	7.4e-08	6.7e-08	9.7e-08	2.3e-07	1.8e-08
Th-229+D	9.6e-08	3.0e-08	5.1e-08	4.6e-08	6.7e-08	1.6e-07	1.3e-08
Th-230	2.0e-08	6.3e-09	1.1e-08	9.7e-09	1.4e-08	3.3e-08	2.6e-09
Th-232	2.3e-08	6.1e-09	1.0e-08	9.4e-09	1.4e-08	3.2e-08	2.6e-09
Pa-231	2.8e-08	7.3e-09	1.2e-08	1.1e-08	1.6e-08	3.8e-08	3.1e-09
U-234	1.6e-08	6.3e-09	1.1e-08	9.7e-09	1.4e-08	3.3e-08	2.6e-09
U-235+D	1.5e-08	5.9e-09	1.0e-08	9.1e-09	1.3e-08	3.1e-08	2.5e-09
U-238+D	1.5e-08	5.6e-09	9.5e-09	8.7e-09	1.3e-08	2.9e-08	2.4e-09
Np-237+D	4.0e-08	1.6e-08	2.7e-08	2.4e-08	3.5e-08	8.1e-08	6.5e-09
Pu-238	3.2e-08	9.1e-09	1.6e-08	1.4e-08	2.0e-08	4.8e-08	3.8e-09
Pu-239	3.3e-08	9.0e-09	1.5e-08	1.4e-08	2.0e-08	4.7e-08	3.8e-09
Pu-240	3.3e-08	9.0e-09	1.5e-08	1.4e-08	2.0e-08	4.7e-08	3.8e-09
Pu-241+D	3.3e-10	7.6e-11	1.3e-10	1.2e-10	1.7e-10	4.0e-10	3.2e-11
Pu-242	3.1e-08	8.5e-09	1.4e-08	1.3e-08	1.9e-08	4.4e-08	3.6e-09
Am-241	4.5e-08	1.7e-08	3.0e-08	2.7e-08	3.9e-08	9.1e-08	7.3e-09
Cm-244	2.9e-08	1.1e-08	1.9e-08	1.7e-08	2.5e-08	5.7e-08	4.6e-09
U-Series	6.5e-08	6.2e-08	1.0e-07	9.5e-08	1.4e-07	1.1e-07	8.5e-09
U-Separ.	3.2e-08	1.2e-08	2.1e-08	1.9e-08	2.7e-08	6.4e-08	5.1e-09
U-Deplet	1.6e-08	6.3e-09	1.1e-08	9.7e-09	1.4e-08	3.3e-08	2.6e-09
Th-Serie	1.4e-07	5.0e-08	8.5e-08	7.8e-08	1.1e-07	2.6e-07	2.1e-08

APPENDIX L

DISCUSSION OF SENSITIVITIES, VARIABILITIES, AND UNCERTAINTIES

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DISCUSSION OF SENSITIVITIES, VARIABILITIES, AND UNCERTAINTIES

1. INTRODUCTION

EPA's evaluation of the potential for recycling scrap metal from nuclear facilities had the following four objectives:

1. Characterize the potential sources of scrap metal that may be available for recycle from nuclear facilities.
2. Estimate the potential normalized annual dose and potential normalized risk to the reasonably maximally exposed individual (RMEI) associated with the release of scrap metal from nuclear facilities.
3. Estimate the potential normalized collective dose and potential normalized collective risk to the exposed population associated with the release of scrap metal.
4. Estimate the minimum detectable concentration (MDC) of radionuclides likely to be contained within or on the surface of scrap metal from nuclear facilities.

Based on the information provided in this Technical Support Document (TSD), as well as the results of the cost-benefit analysis, EPA will decide whether recycling scrap metal from nuclear facilities is viable and whether additional regulatory action is necessary to ensure that release of such materials does not endanger public health and safety.

This appendix discusses the sensitivities and uncertainties in the results reported for each of the four areas of investigation cited above. "Sensitivity" refers to how the results change as a function of changes in fundamental modeling assumptions. "Uncertainties" refer to the uncertainties in the results due to uncertainties in the calculational parameters. A distinction is also made between uncertainties in a single real, but unknown, value (such as the projected time-integrated collective dose to a population) and the variability of a set values from which one value must be selected (such as the annual dose to the RMEI). In the first case, there is a <u>single</u> real, but unknown, value where the uncertainty is due to uncertainty in the calculational parameters used to derive the value. In the latter case, there are <u>many</u> real, but unknown, values, among which one value must be selected which is representative of the set (Hof 94). The distinction between uncertainty in a single real, but unknown, value and the variability among many real, but unknown, values is important in understanding the uncertainties described in this section.

The Agency chose to conduct uncertainty analyses as part of its evaluation of recycling scrap metal from nuclear facilities to ensure its thorough understanding of the adequacies of the data used in its investigations. As stated previously, the results of this evaluation will be used by EPA to determine the potential need for additional regulatory action to address recycle of scrap metal from nuclear facilities. In making this decision, the Agency wants to be aware of uncertainties in the data and their potential effect on the outcome of its decision making process.

Should the Agency choose to initiate a regulatory action, uncertainty analysis becomes even more important. Uncertainty in the data can affect confidence in (1) the degree of protectiveness provided by potential alternative release criteria and (2) the costs and benefits of potential alternative release criteria. When considering various regulatory alternatives and strategies, EPA must understand the significance of the various sources of uncertainty in its analysis and the effects such uncertainty may have on the ultimate calculation of costs and benefits associated with different regulatory options.

The appendix is divided into 5 sections. Sections 2 and 3 address the uncertainties, variabilities, and sensitivities in the tables of values quantifying the characteristics of metal at commercial nuclear power plants and at DOE facilities, as presented in Chapter 4 of the TSD. Sections 4 and 5 characterize the uncertainties, variabilities, and sensitivities in the normalized individual and collective doses presented in Chapters 7 and 9 of the TSD. Section 6 presents the uncertainties, variabilities, and sensitivities in the MDCs described in Chapter 8 of the TSD.

Each section of this appendix is designed to provide the reader with insight into the uncertainty, variability, and sensitivity in the derived estimates of the quantities of metal, the normalized individual and collective doses, and the MDCs. To the extent feasible, quantitative estimates are provided. A more formal quantitative assessment of uncertainties, based on Monte Carlo analyses, is in the planning stage. A full discussion of the Agency's cost-benefit analysis can be found in "Radiation Protection Standards for Scrap Metal: Preliminary Cost-Benefit Analysis (IEC 97)."

2. UNCERTAINTY IN THE CHARACTERISTICS OF POTENTIAL SOURCES OF SCRAP METAL AT COMMERCIAL NUCLEAR POWER PLANTS

Appendix A of this report presents estimates of the quantities of radioactive metal, representing 123 commercial power reactors, that are a potential source of scrap metal following decontamination, decommissioning, and dismantlement of these facilities. The estimates, which are largely based on a model that employed data from two reference facilities, reveal that the total quantity of potentially contaminated metal that may be available for recycling from all commercial nuclear power plants combined is about 500,000 metric tons, of which about 77% is carbon steel, 17% is stainless steel, and the remainder consists of a variety of other metals and alloys. This section presents estimates of the uncertainty and the sources of uncertainty in these overall aggregate values, and the variability among the 123 plants. It demonstrates that the aggregate values are not likely to be higher or lower than a factor of two. However, the variability among individual plants could be more than an order of magnitude.

At best, a model can only approximate real-world conditions. Consequently, all assessment models can be assumed to be inherently inaccurate. The degree of uncertainty in modeled estimates is commonly due to (1) a lack of data for model parameters, (2) improper parameter estimation (parameter bias), (3) improper model formulation (model bias), and/or (4) random or natural variability (parameter variability) that represent stochastic events. Parameter variability has its greatest impact on deterministic models that employ a single value for each parameter to produce a single prediction (estimate).

In summary, the uncertainty of deterministically modeled estimates as provided in this report can be suitably assessed by identifying major differences among current reactors and assessing how well the model characterizes the universe.

2.1 Critical Differences Among U.S. Commercial Reactors

Physical Variations. Nuclear power plants in the United States are by no means standardized. Undoubtedly, the single most important factor affecting the quantity of scrap metal and associated contamination levels is the basic design of the reactor. The two types of U.S. reactors are broadly categorized as pressurized water reactors (PWR) and boiling water reactors (BWR). Of the 123 reactor units, 40 are BWRs manufactured by General Electric and 83 are PWRs manufactured by Westinghouse, Combustion Engineering, or Babcock and Wilcox. Beyond an increase in total steel (and other metals) needed to construct a BWR, another major difference is the larger percentage of steel that is contaminated. This is due to the fact that under normal operating conditions of a BWR, radionuclides enter the steam flow and contaminate turbine plant equipment, which in a PWR are generally assumed uncontaminated.

Besides the designation of a reactor as a BWR or PWR, other important physical variables include the size of the plant and design parameters that reflect the plants' age/period of construction. (The current inventory of reactor units were constructed over a period of about four decades.) Early BWR prototypes, with startup dates in the 1960s, had power ratings of less than 100 MWe, while more recent PWRs were designed to generate between 1,000 and 1,300 MWe. The time period of construction also reflects evolving standards of plant designs and safety, which, in turn, significantly impact scrap metal quantities and contamination levels.

For example, early reactor designs frequently employed run-of-river for cooling, while more recent nuclear facilities employ cooling towers of various designs. Significant quantities of materials inclusive of metals are involved in these alternative cooling systems. Over the years, numerous design modifications also included material changes such as the reduction of Co-59 in alloys or the substitution of zircaloy for stainless steel in reactor vessel internals and primary system components. Such changes not only impact levels of volumetrically distributed activation products but also the levels and buildup of internal and external surface contamination.

Operational Variations. Contamination levels and relative radionuclide composition on interior and exterior metal surfaces are strongly influenced by a host of operational factors. These include (1) years of operation, (2) coolant chemistry and corrosion control, (3) fuel integrity, (4) performance or failures of critical reactor components/systems and their maintenance, and (5) health physics practices and routine cleanup efforts.

Years of Operation. In general, contamination on internal surfaces and inaccessible external surfaces is assumed to accumulate over the 40-year lifespan of a reactor that further assumes 30 effective full-power years (EFPY). Based on the fact that several reactor units have been permanently shutdown well in advance of their 40-year lifespan, it is likely that others among the currently operating units will also face premature shutdown. Conversely, it is also reasonable to conclude that reactors with a demonstrated record of safe operation will be granted an extension beyond the projected 40-year lifespan.

Coolant Chemistry and Corrosion Control. Optimum operation of the chemical volume and control system will have the combined effects of maximizing the removal of radioactive species in the coolant, controlling the production of activation products, and mitigating the formation of radioactive corrosive films on interior surfaces. Radioactivity in corrosion films are dominated by activation products. Activation products are collectively referred to as "crud" and result from the dislocation of small quantities of metals within the primary system to the reactor vessel. When exposed to the high-level neutron flux within the reactor vessel, these metal particulates become radioactive and may be redistributed by the outgoing coolant/steam flow. The process and rate of crud production and buildup of corrosion films on interior surfaces are strongly influenced by additives/contaminants introduced into coolant and the efficiency of on-line cleanup.

Fuel Integrity. Mobile fission products leaked from defective reactor fuel also contribute to internal and external surface contamination. Their concentration (and buildup) are directly related to the number of leaking fuel elements in the reactor core and the time of occurrence within the 40-year lifespan that will represent numerous fuel cycles. The fission products of primary concern with regard to scrap metal include Cs-137 and S-90. While fuel failure is generally not a factor under the control of the reactor's operating staff, early detection and mitigating efforts can minimize the impacts of failed fuel.

System Performance and Maintenance. The performance and efficiency of critical components/systems may have substantial impacts on scrap metal quantities. For example, secondary coolant contamination caused by chronic steam generator tube leakage would significantly impact scrap metal quantities among PWRs. Correspondingly, prompt detection of contaminants in secondary coolant and tube plugging/repair will minimize impacts.

An important factor in plant performance involves preventive maintenance. Contamination is avoided by anticipating/correcting problems before they occur. High maintenance items include seals in pumps and valves that normally contain or isolate radioactive media. Plants with minimum contamination generally incorporate preventive maintenance as part of an expanded routine maintenance program during scheduled outages.

Health Physics Practices. The release of contained contaminants and their spread on exterior surfaces in reactor plant environments are heavily affected by standard health physics practices employed over the life of the plant. For example, the routine use of containment devices can virtually eliminate the release of contaminants during maintenance/repair/replacement of most internally contaminated components. Similarly, the spread of contaminants by plant operating-, maintenance-, and support-personnel is minimized through proper training programs, oversight of work evolutions, and routine health physics surveillance that include decontamination efforts on an ongoing basis.

Decommissioning Alternatives/Schedule. With the publication of NRC's Decommissioning Rule in June 1988, owners and/or operators of licensed nuclear power plants must submit plans for decommissioning their facilities to the NRC for review and approval. Three alternatives that can be used for decommissioning reactor facilities. For the DECON alternative, it is assumed that the owner/operator has a strong incentive to decontaminate and dismantle the retired reactor facility as promptly as possible. Under this option, scrap metal would become available for recycling at about 10 years following permanent shutdown of the reactor.

For the SAFSTOR alternative, a facility may be stabilized and maintained for a period up to 60 years between reactor shutdown and final decommissioning. The obvious impact of a lengthy storage period and associated natural decay on scrap metal is two-fold. For a fraction of contaminated metal inventories with limited starting levels of contamination, natural decay would result in residual levels of contamination approaching background levels and would require no further decontamination at the time of dismantling. For scrap metal with higher starting levels, natural decay will at a minimum reduce the effort required to decontaminate to levels considered suitable for unrestricted/restricted recycling. Thus, SAFSTOR with its attendant natural decay is likely to affect both the quantity of scrap metal available and the required effort for decontamination.

ENTOMB is the third and least likely alternative for decommissioning. This alternative provides for completion of decommissioning beyond 60 years and is likely to be considered only as necessary to protect the public health and safety. It is generally assumed that the period of entombment will be sufficiently long so as to eliminate activity levels in scrap metal to insignificant levels.

Summary. Differences among U.S. reactors that are deemed critical to future quantities of scrap metal involve those that define a facility in terms of its (1) physical design, (2) plant operations, and (3) choice of decommissioning alternatives. A model that adequately reflects the variability of these parameters within the universe is likely to yield estimates that can be viewed with reasonable confidence.

Presented below is a brief discussion of the data that were available for modeling scrap metal estimates and the associated limitations and uncertainties for applying data to the model.

2.2 Reference Models Used to Characterize the Industry

Scope of Data for Reference Facilities. In the 1976-1980 time frame, two extensive studies were conducted for the Nuclear Regulatory Commission by the Pacific Northwest Laboratory to examine the technology, safety, and costs of future decommissioning of large reference nuclear power reactor plants. Because of significant differences between designs, one study selected a large PWR and the other a large BWR to serve as reference reactor facilities. NUREG/CR-0130 (PWR) and NUREG/CR-0672 (BWR) contain detailed information regarding the physical designs and specifications of major reactor components and derived best estimates of residual radio-activity levels based on empirical dose rate measurements.

With the publication of the NRC's Decommissioning Rule on June 27, 1988 that required owners/operators of nuclear reactor facilities to submit decommissioning plans, earlier NUREG reports were revised to reflect changes in cost. Technical data contained in revised NUREG/CR-5884 (PWR) and NUREG/CR-6174 (BWR), however, remained unchanged. To date, these two studies represent the principal available studies for defining a Reference PWR and BWR.

In spite of the immense data contained in these two studies, the information is by no means complete. While data for contaminated steel are sufficiently detailed and considered highly reliable, no information exists regarding inventories and contamination levels for metals other than steel. Estimates for scrap metal involving other metals or metal alloys cited for Reference facilities in this report were based on inference that employed reasonable, but unconfirmed, assumptions.

From a single study (Bryan and Dudley 1974) that identified total plant inventories of galvanized iron, copper, inconel, lead, bronze, aluminum, brass, nickel, and silver, estimates of contaminated metal inventories were derived by assuming that the contaminated fractions among total plant inventories, for each of the above-cited metals, parallels the contaminated fraction of carbon steel for Reference BWR and Reference PWR.

The validity of Reference facility scrap metal estimates for metals other than steel is further obscured by the fact that sizable (but undefined) fractions of some metals may not be retrievable or exist in something other than its elemental form. In summary, while steel inventories for Reference facilities are adequately defined, reasonable, but unconfirmed, assumptions were used to provide best estimates for contaminated quantities of other metals and metal alloys.

Industry Estimates that Account for Period of Construction and Plant Size. The accident at the Unit-2 reactor at the Three Mile Island Nuclear Station in March 1979 was the major impetus for revisions in reactor design, operations, and maintenance. It is important, therefore, that basic characteristics of the reference sites are representative of the total reactor units. Table L.2-1 provides an overview of the distribution of U.S. reactors with regard to this important landmark in time.

Table L.2-1. Start-up Data of U.S. Reactors Relative to the TMI Accident

	BWR (%)	PWR (%)	All (%)
Pre-TMI	62	38	54
Post-TMI	50	50	46

The 1,175 MWe Trojan Nuclear Plant designed by Westinghouse began commercial operation in 1976 and employs standard cooling towers. Its basic pre-TMI design represents a period of construction that is midway within the four decades of plant construction. Considered a typical pressurized water reactor in the original 1978 study (NUREG/CR-0130), it is still regarded as such today (NUREG/CR-5884).

Washington Nuclear Project-2 facility began operation in the post-TMI-Accident era in 1984. Constructed by General Electric, its 1,155 MWe design, which includes forced draft cooling towers, is likewise regarded today as the designated Reference BWR facility (NUREG/CR-6174).

For estimating total industry inventories, a scaling factor was applied to the balance of 121 reactor units to account for differences in plant size. The scaling factor used in this report was one that has been recommended and used by the DOE (DOE 1995). Empirically derived from reactor mass data and power output, the scaling factor assumes that all metal inventories (contaminated and uncontaminated) for both PWRs and BWRs can be correlated to the corresponding Reference plant in proportion to the design power rating to the 2/3 power ($MWe^{2/3}$).

Validation of this scaling factor and its application to reactors ranging from 50 MWe to nearly 1,300 MWe, however, has not been documented and, therefore, represents an undefined source of uncertainty in modeled scrap metal quantities for the industry as a whole.

Accounting for Operational and Intrinsic Factors

Numerous operational factors in combination with previously described intrinsic factors will undoubtedly have some effect on scrap metal quantities; but their primary effect will be their impact on the levels of scrap metal contamination. Thus, while it is a virtual guarantee that systems in contact with primary coolant, liquid radioactive waste, and BWR-steam will become contaminated, the level of contamination for a given system is likely to vary over several orders of magnitude among reactor plants.

This wide range of activity levels reflects the combined effects of episodic equipment failure, fuel leakage, and routine operational practices over the plant's 40-year lifespan. Reference facility study data provided credible information regarding the relative radionuclide distribution as well as contamination estimates that were based on empirical dose rate measurements. However, these "one-time" measurements (in time and space) provided no information about the variability that must be anticipated among individual facilities.

The anticipated wide range of values regarding radionuclide composition and absolute quantities of residual contamination are shown in Table A3-6 of Appendix A and are reproduced here for illustration purposes. This data set represents measurements for three BWRs and three PWRs (NUREG/CR-4289). The two most abundant radionuclides were Fe-55 and Co-60 in all cases except Monticello. These two radionuclides constituted over 95% of the estimated inventories at Humboldt Bay. At Indian Point Unit One, Dresden Unit One, Turkey Point Unit Three, and Rancho Seco, they accounted for 82, 74, 55, and 46%, respectively, of the total estimated inventory. Although Fe-55 and Co-60 accounted for the majority of the inventory, the relationship between the two radionuclides was quite variable. The ratio of Fe-55 and Co-60 at the six generating stations ranged from 15 to 1 at Humboldt Bay to 0.01 to 1 at Monticello. Zinc-65 constituted 84% of the total inventory at Monticello. This large variability was presumably due to differences in the water chemistry, which controlled the corrosion and deposition of these radionuclides, and differences in operating history, which affect the production ratios since the radionuclides have an approximate factor of two difference in half-life.

The largest ranges, as a percentage of the total inventory, were noted for Zn-65 and Ni-63, which ranged from 0.09 to 84 % and 0.04 to 19%, respectively. This wide range was related to the composition of the materials of construction used in the primary systems of the reactors. The large amounts of Zn-65 observed at Monticello, Dresden Unit One, and Indian Point Unit One were the result of the use of admiralty brass heat exchangers (29% zinc).

Table A3-6 of Appendix A. Radionuclide Composition of Internal Surface Contamination*

Radionuclide	Composition in Percent of Total Activity Decay Corrected to Shutdown Date					
	BWRs			PWRs		
	Humboldt Bay	Dresden-1	Monticello	Indian Point-1	Turkey Point-3	Rancho Seco
Mn-54	3	0.9	1	4	0.4	4
Fe-55	90	28	1	67	31	28
Co-57	—	—	—	—	43	24
Co-60	6	46	11	15	24	18
Ni-59	—	0.09	—	0.02	0.004	0.1
Ni-63	0.2	5	0.04	2	0.1	19
Zn-65	—	19	84	11	1	0.09
Sr-90	0.004	0.007	0.002	0.0007	0.0008	< 0.01
Nb-94	< 0.004	< 0.003	< 0.1	0.0008	< 0.004	< 0.004
Tc-99	3×10^{-4}	4×10^{-5}	8×10^{-5}	8×10^{-5}	0.008	< 0.005
Ag-110m	—	—	—	—	—	4
I-129	$< 3 \times 10^{-6}$	$< 1 \times 10^{-5}$	$< 1 \times 10^{-6}$	2×10^{-5}	< 0.003	$< 1 \times 10^{-5}$
Cs-137	0.5	0.04	2	0.5	—	0.4
Ce-144	—	1	—	—	0.2	< 0.04
TRU**	0.005	0.1	0.008	0.002	0.006	0.001
Total Plant Inventory (Curies)	596	2,350	448	1,070	2,580	4,460

Source: NUREG/CR-4289

* Excludes highly activated metal components of the reactor pressure vessel and internals, and activated concretes.

** Transuranic alpha-emitting radionuclides with half-lives greater than 5 years, include Pu-238, Pu-239, Pu-240, Am-241, Am-243, and Cm-244.

The large component of Ni-63 in the Rancho Seco inventory was due to the more extensive use of inconel (80% nickel) in the primary system of this reactor. The low percentage of Ni-63 at Monticello was typical of newer BWRs, which make minimal use of nickel alloys since they are subject to higher rates of corrosion in the relatively more oxidizing environment of the BWR primary coolant loop. Excluding Rancho Seco and Monticello, the relative abundance of Ni-63 ranged a factor of 50, from 0.1% of the total inventory at Turkey Point Unit-3 to 5% at Dresden Unit One.

To reflect the large variations reported in NUREG/CR-4289, draft NUREG/CR-1496, and currently available decommissioning plans, the following three levels of contamination were defined for characterizing individual reactor systems for all BWRs and PWRs.

Low Contamination: $< 1 \times 10^5$ dpm/100 cm^2
Medium Contamination: 1×10^5 to 1×10^7 dpm/100 cm^2
High Contamination: $> 1 \times 10^7$ dpm/100 cm^2

For example, in Table 5-2B of Appendix A, the Standby Gas Treatment System in a BWR was identified to most likely exhibit Medium Contamination levels. This can be interpreted to imply that most (if not all) of the 40 Standby Gas Treatment Systems for the 40 BWR's will fall within the range of 1×10^5 dpm/100 cm^2 and 1×10^7 dpm/100 cm^2.

While current data are insufficient to make further assumptions regarding the likely distribution of values within the assigned range of contamination, it would appear reasonable to conclude that the average contamination for the 40 BWR Standby Gas Treatment Systems would be central to the range of about 1×10^6 dpm/100 cm^2.

Projected Choices Among Decommissioning Alternatives. Currently, only a few licensees have submitted decommissioning plans to the NRC for review and approval. Moreover, a factor confounding this limited data base is that these facilities did not experience normal operations and/or operated for the expected 40 years of plant life. Therefore, preliminary choices of decommissioning alternatives by these few facilities do not provide a credible basis for projecting decommissioning alternatives for the vast majority of current reactor units.

In this report, it was conservatively assumed that dismantling and release of scrap metal for all reactor units will occur at the earliest possible time or about 10 years post-shutdown following a standard 40-year period of plant operation.

Like operational factors, the choice of decommissioning alternatives is also likely to have some impact on scrap metal quantities but more importantly will affect residual contamination levels. Since the residual radionuclide contamination is a mixture with varying half-lives, the relative composition of the radionuclides present will change, along with the decrease in absolute concentrations, with time after shutdown. Initially, significant radionuclides such as Co-60, Fe-55, Co-58, and Zn-65 will decay rapidly in comparison to radionuclides with longer half-lives (e.g., Ni-63, Sr-90, and Cs-137).

Residual quantities at 10, 30, and 50 years post-shutdown in corrosion films and in primary coolant for Reference PWR and Reference BWR were cited in Appendix A and reflect between 10- and 100-fold reductions in residual contamination. It should be noted that for systems contaminated by media other than primary coolant (e.g., radioactive waste, fuel pool) the changes in radionuclide inventories with time will be less dramatic since concentrations of longer-lived radionuclides, inclusive of Cs-137 and Sr-90, will be enhanced in these media.

2.3 Summary Conclusions

Diversity among the current U.S. inventory of 123 licensed nuclear reactors is likely to yield variable quantities of scrap metal among individual reactor units at the time of decommissioning. The quantities and types of metal scrap and their levels of residual contamination will be influenced by numerous factors.

Foremost in defining potential scrap metal quantities are physical parameters that are determined by the class of reactor, reactor size, and period of construction. These physical variables are well documented for the 123 reactor units and were factored into modeled scrap metal estimates by (1) employing a Reference plant for each of the two major reactor types, (2) use of a empirical scaling factor where power rating served as a surrogate measure of reactor size, and (3) use of Reference facilities that were constructed about midway through the 30-40 year construction period that defines the nuclear power industry. (Moreover, one Reference facility was pre-TMI-2 era and the other was post-TMI-2 era.)

Secondary parameters that are likely to indirectly influence the quantities of scrap metal released for recycling are those that impact the levels of contamination. The ability to release scrap metal assumes the cost-effective decontamination of scrap metal to levels below prescribed limits. In general, increasing levels of contamination are likely to yield decreasing percentages of scrap metal available for recycling.

To a large extent, variations in radionuclide composition, distribution, and absolute concentrations are the result of operational factors. Operational factors, however, are not easily incorporated into modeled estimates and were, therefore, not considered. In part, this is due to the probabilistic/episodic occurrence of some operational factors (e.g., system/ component failure, fuel leakage) and the subjective nature of others (e.g., quality of coolant water chemistry, corrosion control, health physics practices, etc.)

To reflect the high degree of variability as reported by earlier studies and a small number of decommissioning plans, plant systems in this report were grouped into one of three levels of contamination, where each level represents a range of values that spans three or more orders of magnitude.

It should be noted, however, that the projected contamination levels as suggested in this report may very well represent upper-bound values. This is due to the biased data from which modeled data were derived. Past studies (NUREG/CR-4289) and current decommissioning plans represent reactor facilities with abnormal histories of operation and are not likely to be considered representative of the industry at large. At a minimum, the bulk of reactor operations at these facilities preceded the 1979 TMI-2 accident and reflect material composition, plant systems, and operational standards of the pre-TMI era. The accident triggered major reforms in the commercial nuclear industry in the form of more stringent Federal regulations and performance standards issued by the NRC. Post-TMI reforms also reflect the introduction of new standards, guidance, recommendations, and good practices issued by the American National Standard Institute (ANSI), American Nuclear Society (ANS), American Society for Testing and Materials (ASTM), National Council on Radiation Protection (NCRP), Electric Power Research Institute (EPRI), and others. By far, the single most important of these organizations to influence post-TMI plant operations is the Institute of Nuclear Power Operations (INPO). Collectively, their efforts to improve and standardize reactor plant operations can be expected to have a dual effect on contamination levels at the time of decommissioning: (1) on average, contamination levels can be expected to be below those identified in this report, and (2) the range or variability of contamination levels among individual plants are likely to diminish.

The final variable affecting scrap metal contamination levels (and, to a lesser extent, scrap metal quantities) is the choice of decommissioning alternatives. SAFSTOR with its extended delay in dismantling/decommissioning has the obvious impact of reducing contamination levels by up to several orders of magnitude.

Depending on prevailing decontamination technologies and economic factors, a reduction in residual contamination levels in scrap could significantly increase scrap metal quantities. For example, if prevailing cost-effective decontamination technologies were limited to reducing contamination to four orders of magnitude, scrap metal at 10-years post-shutdown with activity levels > 5×10^7 dpm/100 cm^2 could not be expected to meet the current release standard of 5,000 dpm/100 cm^2 and would, therefore, be excluded from recycling. Under the SAFSTOR alternative, if through natural decay starting contamination levels were reduced by several orders of magnitude, an expanded fraction of the total pool of scrap metal can be expected to meet the release criteria of a prevailing standard.

At this time, however, the vast majority of reactor licensees have not revealed their preference for a specific decommissioning alternative and speculation regarding decontamination technologies for nearly a century into the future would be unwise. For these reasons, uncertainties associated with decommissioning alternatives were not addressed in this report. Scrap metal quantities and residual contamination levels were based on a 10-year post-reactor shutdown period and current decontamination technologies.

In conclusion, model parameters that reflect power plant operations and decommissioning alternatives can not be adequately defined at this time. Although their exclusion from the model prevents a rigorous quantitative analysis regarding the uncertainty of scrap metal estimates presented in this report, current data are, nevertheless, sufficient to support the following statements:

1. Scrap metal quantities and levels of contamination will vary considerably among individual plants.
2. Physical differences inclusive of plant design, power rating, and period of construction are thought to be the most important parameters affecting scrap metal quantities for individual reactors and were incorporated into the modeled results.
3. Parameters that could not be readily defined (i.e., operational factors) are likely to represent a continuum with a symmetrical distribution about a mean value. Thus, factors contributing to low quantities of scrap metal containing radioactive contamination for some plants will be offset by others yielding higher than expected scrap metal quantities. As such, the uncertainty in the collective quantities and radionuclide inventories for all plants combined are likely to be considerably smaller than the variability among plants.

4. Variations among reactor plants pertaining to operational factors and the selection of a decommissioning alternative are more likely to impact contamination levels of individual reactor systems as opposed to the mass quantity of radioactive metal.

5. Based on currently available information, it is concluded that the collective industry inventory of scrap metal potentially available for recycling, as estimated in this report, is not likely to vary by more than a factor of two.

3. UNCERTAINTY IN THE CHARACTERISTICS OF POTENTIAL SOURCES OF SCRAP METAL FROM DOE FACILITIES

This section reviews the limitations of available data, identifies underlying assumptions that were employed in deriving scrap metal estimates, and provides a subjective interpretation of their potential impacts on uncertainty. In general, the section demonstrates that the uncertainty in the estimate of the existing inventory of potentially contaminated scrap metal at DOE facilities of 171,089 MT is small, i.e., a factor of 2 or less. However, the uncertainty in the estimate of the total future quantity of DOE scrap metal that will be generated following decommissioning of DOE facilities (i.e., 925,614 MT) is relatively large, approximately a factor of 2 or greater, and probably larger than the estimated value.

Section 4.1 of Chapter 4 provides best estimates of scrap metal quantities that are currently stored at various DOE facilities and projected scrap metal quantities that will become available at some future date. Projected scrap metal quantities are linked in time and quantity to the schedule and scope of future decommissioning activities of the DOE Nuclear Weapons Complex.

Under ideal conditions, available data would have provided complete information for each DOE facility. On the basis of empirical measurements, such information would (1) identify total scrap metal quantities, (2) define contributing percentages by metal type, (3) categorize scrap metal by physical form, and (4) characterize radioactive contamination by identifying dominant radionuclides and their relative abundance.

Estimates presented in the TSD, however, were based on data that were frequently speculative, incomplete, and in other cases insufficiently detailed. Quantitative and qualitative gaps in data, therefore, necessitated the use of surrogate values, assumptions, and interpolation.

3.1 Review of Primary Data Sources and Data Selection Criteria

Scrap metal estimates were largely taken or derived from data presented in four source documents that included the following:

- DOE's 1996 Material in Inventory (MIN) Report (MIN 96).
- HAZWRAP's 1995 Scrap Metal Inventory Report (HAZ 95).
- EPA's 1995 Contract Report by SC&A, Inc. (SCA 95).

- DOE's 1995 Decontamination and Decommissioning Report for Gaseous Diffusion Facilities (DOE 95a).

In general, data contained in these reports were either identical (or in close agreement); at other times, a given report contained unique data that complemented the other reports. In a few instances, differences existed in reported values that required resolution.

For data selection and data resolution, the following criteria were employed:

- Scrap data contained in MIN 96 were taken as most current and, therefore, considered most reliable.
- Unless scrap metal was explicitly specified as contaminated, all unspecified scrap metal was assumed to be 88% contaminated and 12% clean. (Note: Clean metal was not considered radioactive scrap metal and was, therefore, excluded for consideration in EPA's analysis.)
- For existing scrap metal quantities, the breakdown by metal types and physical forms was exclusively based on data contained in HAZ 95.
- For future scrap metal quantities, the breakdown by metal types and physical form was based on current scrap metal data.
- The percent of future scrap metal likely to be contaminated (i.e., scrap metal) was assumed to be the same as the current percentage of 88%.

In summary, the combination of data contained in the four documents provided the bases for scrap metal estimates representing 13 DOE facilities. These facilities are considered principal sources of existing and future scrap metal that may be suitable for recycling. Table L.4-1 identifies site-specific scrap metal quantities and the source document from which data were obtained.

Table L.3-1. Selection of Data Sources for Scrap Metal Quantities at DOE Facilities

DOE Site	Existing scrap metal (in MTs)		Future scrap metal (in MTs)		
	Source Document		Source Document		
	MIN 96	HAZ 95	MIN 96	EPA/SCA 95	DOE 95
Fernald	4,218			135,623	
Hanford	377			91,798	
INEL	727		33,486		
LANL		3,099		2,686	
NTS	264		—	—	—
ORNL	1,129		—	—	—
Y-12	9,065		—	—	—
K-25	29,357				212,706
Paducah	48,374				230,886
Portsmouth	8,914				189,072
Rocky Flats		24,543		26,303	
SRS	13,183		3,054		
Weldon Spring		27,839	—	—	—
SubTotal	171,089		925,614		
TOTAL	1,096,703				

3.2 Uncertainties Pertaining to Existing Scrap Metal Quantities

Table L.4-1 identifies MIN 96 and HAZ 95 Reports as principal sources for characterizing existing scrap metal sources. The information, however, is limited to deterministic (i.e., single) values of scrap metal estimates at individual DOE sites; no additional data are provided that would further define the degree of accuracy or variability of cited values. In acknowledgement of these limitations, the MIN 96 Report stated the following:

> "... Because of limited data, this report does not attempt to capture the exact amount of each material in inventory. Rather, it attempts to capture the general magnitude of the inventory of each material."

Elsewhere, the MIN 96 Report concluded that while the ". . . Department maintains detailed inventory systems of weapons components . . . there is no reliable system to identify a complete inventory of scrap metal and equipment." (Emphasis added)

A reasonable interpretation of these statements is that cited scrap metal quantities reflect best estimates (as opposed to comprehensive measurements) and, therefore, pose a significant but undefined level of uncertainty. As a rule, deterministic data preclude a rigorous approach for assessing uncertainty. Assessing the reliability of deterministic data, therefore, is restricted to a subjective evaluation that focuses on the methods employed for data collection.

DOE Data Collection Methods. DOE's 1996 Materials in Inventory Initiative was a year-long Department-wide effort aimed to improve management and disposition for materials that may no longer be needed. The objectives of this effort focused on management approaches for (1) uncontaminated materials, (2) suspect materials, (3) contaminated material, and (4) clearing suspect or contaminated materials to the property management system for reuse or release. Data sources for the MIN Scrap Metal and Equipment Team that developed scrap metal estimates included the following:

- DOE Regulations, Policies, and Orders that served as sources of information on requirements and procedures for managing scrap metal and equipment.
- Studies conducted within the past five years pertaining to scrap metal inventories within the DOE complex.
- Information collected in response to surveys, site-visits, and national conferences.

Useable data in MIN 96, however, were only defined for 10 of the 13 sites that had previously been identified as significant scrap metal sources. Furthermore, data collection methods for individual sites varied; while some sites have developed and maintain databases that are current for scrap metal inventories, others relied exclusively on historical knowledge to determine quantities of scrap under their control.

With regard to the reliability of collected data, the MIN Scrap Metal and Equipment Team offered the following statement:

> " . . . Data limitations include the following: (1) no information was received on either scrap or equipment for [several DOE facilities] . . . (2) some information was submitted in summary form only, without site-specific breakouts; (3) some sites supplied complete information on some topics and partial or no information on others . . . and (4) some data could not be tabulated because it was descriptive rather than quantitative or expressed in units inconsistent with the units used in this report and could not be readily converted."

Noteworthy is the MIN 96 Report's reference to "partial or no information" that pertains to the fact that only about one-fifth of the total scrap metal inventory has been assessed for the presence of radioactive contamination. In other words, four-fifths (or 80%) of existing metal inventories have not been assessed for radioactive contamination. Of the assessed 0.2 fraction, about 88% was determined to be radioactively contaminated. This relationship was used to estimate the percentage of contaminated scrap within the unassessed 0.8 fraction of metal scrap.

In summary, the collective uncertainty of derived scrap metal quantities reflects the combination of uncertainties contributed by the following:

(1) the questionable accuracy of total scrap metal estimates as reported by individual DOE sites that in some instances were solely based on historical records;

(2) the large percentage of scrap (~80%) that was "undetermined" with regard to radioactive contamination and the resultant need to apply a scaling factor derived from the 20% of scrap that had been assessed for contamination; and

(3) the variability of existing scrap metal inventories as a function of time.

With regard to the third component of uncertainty, most sites reporting data for the MIN initiative indicated that their inventories of existing scrap may be sold or otherwise dispositioned on a routine basis. The extent of variation in inventories with time can, therefore, not be assessed from the snapshot of inventories as currently reported.

3.3 Comparison of Current Estimates with Past Study Data

For a variety of reasons, other studies have been conducted over the years that have estimated DOE scrap metal inventories. Findings of these reports are briefly summarized below to offer a sense of perspective of current data presented in the TSD.

In 1991, DOE developed a white paper that discussed the possibility of recycling radioactively contaminated scrap metal (DOE 91). The white paper assessed scrap metal inventories at seven sites: The Nevada Test Site, Oak Ridge National Lab, K-25 and Y-12 at Oak Ridge, the Paducah and Portsmouth GDPs, and Fernald. Moreover, the scope of the white paper included scrap metal that had been buried in the past, primarily at the Nevada Test Site. The white paper was updated in 1992, with an estimate of approximately 1.5 million metric tons of radioactively-contaminated scrap metal at the seven sites. This estimate included scrap metal buried at NTS, which accounted for approximately 80% of the total estimated scrap metal inventory (Table L.3-2).

In 1993, the Quadrex Corporation conducted a study of scrap metal in the DOE weapons complex that also focused on scrap metal estimates. The Quadrex study found approximately 396,000 tons of scrap metal in inventory, including an undefined quantity of buried scrap at NTS (QUA 93).

In 1994, DOE initiated development of a Baseline Inventory Report for materials held by the Department. The data gathering effort supporting the Baseline Inventory Report focused on existing inventories. The reported data included a category termed "scrap" that represented more than 40 different classes of materials. The classifications of this "scrap" category included materials ranging from aluminum to wood, but also included "miscellaneous," "multiple," "scrap," and "uncontaminated" classifications that did not clearly distinguish component materials. Of the 40 classifications, 17 were unambiguously identifiable as metals (exclusive of heavy metals) and were reported by weight. The weights reported for these 17 classifications within the "scrap" category totaled 225,242 tons (DOE 95b).

In 1995, Parsons Engineering Science and others produced the final report of a scrap metal inventory conducted at the following sites in the DOE complex. This study, referred to as the HAZWRAP Report, included: Oak Ridge National Lab, the K-25 and Y-12 sites at Oak Ridge, the Paducah and Portsmouth gaseous diffusion plants, Hanford, Idaho National Engineering Laboratory, Los Alamos National Laboratory, Rocky Flats, Savannah River, Pinellas, Fernald, and Weldon Spring. (Note: As identified in Table L.3-1, HAZWRAP data were utilized for only three DOE sites.) This collective estimate of 202,869 metric tons, however, did not distinguish between clean and contaminated scrap metal.

Table L.3-2. Comparison of Past with Current Estimates of Scrap Metal Inventories (MT).

Metal Type	1992 Update DOE White Paper		1992 Quadrex Report	1995 Baseline Inventory	1995 Parsons Study (HAZWRAP)	Current (1997) TSD
	Buried	Stored	Stored & Buried	Stored Only	Stored Only	Stored Only
Aluminum	129,000	32,000	16,250	6,588	6,810	7,504
Brass	—	—	10	—	—	—
Copper	25,900	6,500	11,215	4,631	4,233	—
Copper & Brass	—	—	—	125	—	1,679
Monel	—	—	1,745	377	—	—
Nickel	163,300	40,850	47,524	9,700	9,700	11,716
Steel	875,000	218,800	143,221	37,903	—	—
Carbon Steel	—	—	—	122,183	157,502	136,974
Stainless Steel	—	—	—	24,757	24,587	6,665
Tin & Iron	—	—	—	9,677	—	—
Other/Misc.	—	—	175,594	9,301	37	6,551
TOTAL	1,193,200	298,150	395,559	225,242	202,869	171,089

A rigid comparison of data reported by past studies with current estimates is inappropriate due the impacts of time and differences in the scope among these studies that include: (1) the total number of DOE sites assessed, (2) the inclusion of buried scrap, and (3) the failure to differentiate contaminated scrap metal from total scrap metal inventories. In spite of these acknowledged differences, current estimates nevertheless appear "consistent" with past estimates.

On the basis of available data, it is, therefore, concluded that the current estimated value of 171,000 metric tons of contaminated scrap is <u>not</u> likely to differ from the true value by more than a factor of two (2). Thus, a lower- and upper-bound value of existing contaminated scrap metal is defined by 85,500 and 342,000 metric tons.

3.4 Uncertainties Regarding Future Quantities of Scrap Metal

Section 4.1.3 of Chapter 4 identified MIN 96, DOE 95, and EPA's 1995 Contract Report as the principal sources for estimating future scrap quantities. Of the 13 sites with existing scrap metal inventories, however, only nine sites were identified as future sources of scrap metal estimated at about 925,000 metric tons.

For individual DOE sites, point estimates were largely derived from historical data pertaining to design specifications of buildings, structures, and process equipment that have been slated for decommissioning.

The level of uncertainty regarding future quantities of scrap metal is undoubtedly higher than that of existing scrap metal quantities. Compounding the shared uncertainty of simply quantifying a known aggregate of metal components is the incomplete and dubious decommissioning schedule on which future scrap metal estimates are based.

Rigid assumptions regarding future political, social, and economic factors that may significantly impact the current decommissioning schedule cannot readily be factored into a discussion of uncertainty. Clearly, a reduced scope of decommissioning activity is likely to yield future scrap metal quantities below the estimated value of 925,000 metric tons. Conversely, an expanded decommissioning that extends beyond the nine DOE sites defined in this report would be expected to significantly raise the current estimate of projected scrap quantities.

The potential for underestimating future scrap quantities was in fact raised in the MIN 96 Report in the following statement:

> "... The 1995 Sanford Cohen study for EPA [cited in the report as the EPA 95 Contract Report] projected the quantity of scrap metal to be generated from future decommissioning as 1.06 million tons[1] ... [which] ... is believed to substantially underestimate the total scrap metal resulting from future decommissioning."

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[1] 1.06 million tons is equal to about 963,000 metric tonnes.

3.5 Uncertainties Regarding Metal Type and Physical Form

Characterization of existing scrap inventories by metal type and form is provided in both the MIN 96 and HAZ 95 Reports. A significant difference between these reports is that MIN 96 data was limited to the collective contributions by metal type on a DOE-wide basis while HAZ 95 provided data on a site-specific basis.

Identification of specific scrap metal components (i.e., physical form) for existing scrap inventories is limited to the HAZWRAP study, which was provided for select DOE sites. Table L.3-3 identifies the availability of data pertaining to the contribution of metal types to scrap inventories and their physical form. Site-specific data was available for 10 DOE sites and was reported for the full inventory of scrap metal (as opposed to a subset or sample). For three sites (i.e., Hanford, INEL, and NTS), DOE-wide values cited in MIN 96 were used as surrogate estimates.

For future scrap metal, characterization by metal type and physical form was limited to the EPA 95 Contract Report that contained estimates for only 4 of 13 DOE sites.

For existing scrap metal inventories, the large number of reporting sites (i.e., 10 out of 13) and their majority contribution (i.e., > 95%) to the total DOE scrap inventory assure a near absolute certainty of values as reported in HAZ 95 and adopted in the TSD.

For future scrap, available data for characterizing inventories by metal type and form were considered inadequate and were, therefore, forfeited. The characterization profile of existing scrap was considered the preferred option for defining future scrap. Thus, the uncertainty regarding metal type and physical form of future scrap, as defined in this report, is dictated by the degree of similarity (or dissimilarity) between present scrap inventories and future scrap inventories.

Future scrap that will be derived from decommissioning activity will undoubtedly be different from scrap currently stored at DOE sites. These differences, however, are not likely to be profound.

Table L.3-3. Characterization of Existing and Future Scrap by Metal Type and Physical Form

DOE Site	Existing Scrap		Future Scrap	
	Metal Type Data	Physical Form Data	Metal Type Data	Physical Form Data
Fernald	HAZ95 Site Specific	HAZ95 Site Specific		
Hanford	MIN96 DOE Average		EPA95 Site Specific	EPA95 Site Specific
INEL	MIN96 DOE Average		EPA95 Site Specific	EPA95 Site Specific
LANL	HAZ95 Site Specific	HAZ95 Site Specific		
NTS	MIN96 DOE Average			
ORNL	HAZ95 Site Specific	HAZ95 Site Specific		
Y-12	HAZ95 Site Specific	HAZ95 Site Specific		
K-25	HAZ95 Site Specific	HAZ95 Site Specific		
Paducah	HAZ95 Site Specific	HAZ95 Site Specific		
Portsmouth	HAZ95 Site Specific	HAZ95 Site Specific		
Rocky Flats	HAZ95 Site Specific	HAZ95 Site Specific	EPA95 Site Specific	EPA95 Site Specific
SRS	HAZ95 Site Specific	HAZ95 Site Specific	MIN96 Site Specific	MIN96 Site Specific
Weldon Spring	HAZ95 Site Specific	HAZ95 Site Specific		

4. VARIABILITY, UNCERTAINTY AND SENSITIVITY OF THE NORMALIZED DOSES AND RISKS TO THE RMEI

Table 7-1 presents the derived doses and risks to the RMEI for 40 radionuclides and four nuclide combinations, normalized to a specific activity of 1 pCi/g in scrap of each nuclide (or parent nuclide in the case of a radioactive decay series). This section is concerned with the variability, uncertainty and sensitivity of the reported normalized doses and risks, and their potential significance. In general, the analyses demonstrate that the normalized doses to the RMEI from a given nuclide could, in extreme cases, be higher by a factor of 5 to 50, or lower by a factor of 100 to 500. The uncertainty in the normalized risks are similar, except that, depending on the shape of the dose response curve for very low doses and dose rates, the risks could be zero in such cases.

4.1 Variability in Normalized Individual Doses

At some time in the future, nuclear facilities may begin to release scrap metal for recycling. The flow of the metal will take the form of perhaps thousands of truck and rail shipments to scrap dealers and to the approximate 130 mills that currently recycle scrap metal. The flow will continue for several decades until the backlog of scrap metal and scrap generated during decommissioning of nuclear facilities is disposed of.

In any given year, the number of loads shipped to a given mill is expected to be highly variable, with some mills potentially receiving a significant fraction of their feedstock from nuclear facilities. A simplified view of the initial processing of the scrap is depicted in Figure 5-1. Section 5.2 lists 17 distinct but simplified exposure scenarios, each of which is used to model the exposure of one or more individuals. In reality, each shipment is unique, each mill is unique, each transport route is unique, each end use of the steel, slag and baghouse dust is unique, and each exposed individual is unique in terms of physical characteristics, behavior patterns and radiosensitivity.

Let us assume that the residual radionuclide concentrations in the scrap could be accurately determined. Let us further assume that the radiation exposure of each of the perhaps thousands of people that may come into contact or close proximity with the scrap metal and with the various products and side streams produced as a result of recycling were closely monitored and that the monitoring devices were able to measure extremely small doses above background, so that the incremental annual doses to each individual due to recycling were known precisely. The distribution of these doses could be used to assess the variability of the normalized annual doses and risks associated with the free release of the scrap metal. There is little doubt that these doses would vary widely among individuals and as a function of time.

If the individuals were grouped according to the exposure scenarios listed in Table 5-1, an estimate could be made of the range of doses of the individuals in each group. In effect, after the recycling has occurred and after the monitoring was performed, the results would present a measure of the variability of the normalized doses and risks for each group and to each individual in that group as a function of time.

Of course, empirical measurements of the doses to these individuals do not actually exist. However, the concept still holds. Through the use of models, we estimate the range of plausible doses that may be experienced by the individuals in each exposure category if, in fact, scrap metal from nuclear facilities were released.

4.2 Uncertainty in Normalized Individual Doses

In the above hypothetical example, in addition to variability in the doses among individuals and groups of individuals, there is uncertainty in the dose or risk to any one of the monitored individuals. Hence, after selecting the RMEI among the population of exposed individuals, the question is, how uncertain is the estimated normalized dose or risk for those individuals?

Since we have postulated a set of exposures conditions used to model each exposure scenario, the only uncertainty is in calculating the radiation field in the case of external exposure, and in converting both external and internal exposures to dose or risk.

The conversion of exposure to dose involves converting external exposure in units of Roentgens/hr (R/hr) to effective dose equivalent (EDE) in mrem/y, and converting radionuclide inhalation and ingestion rates, in units of pCi inhaled or ingested per year, to committed effective dose equivalent (CEDE), also in units of mrem/y. Given the average annual external exposure in R/hr to which an individual is exposed, the conversion of exposure to EDE is associated with little uncertainty as compared to the variability in the exposures. Specifically, a defined external radiation field in units of R/hr is converted to mrem/hr using widely accepted procedures. The procedures are based on certain assumptions regarding the attenuation and absorption of the radiation as it passes through the body. In fact, for the same radiation field, the pattern of energy deposition within the body will vary depending on the orientation of the field (i.e., is the field created by a source above, below, behind, in front of, or surrounding the individual?) and also among individuals; e.g., larger people will receive a smaller dose than smaller people due to self-shielding by the body. The assumed orientation is explicitly accounted for in each external exposure scenario (see discussions in Appendix H). The differences in the effective dose are small, typically varying by less than a factor of two.

The conversion of a given intake of a radionuclide by inhalation or ingestion to CEDE is associated with a considerable degree of uncertainty because the dose conversion factors are based on standardized assumptions that apply to a Reference Man, defined to be representative of a typical adult. Due to individual variability, the dose conversion factors could result in either a several fold overestimate or underestimate of the dose.

The conversion of exposure to risk of cancer incidence using slope factors (expressed in units of lifetime risk of cancer per unit intake of a given radionuclide) is associated with a relatively high degree of uncertainty. The slope factors used to derive risks are based primarily on extrapolations from epidemiological data where the individuals were exposed to doses many orders of magnitude higher than the exposures that may be associated with the free release of scrap metal from nuclear facilities. EPA 96 discusses these uncertainties. In summary, it is unlikely that the actual slope factors could be higher than the derived slope factors by more than a factor of two or three. However, there is also a possibility that the slope factors at these very low doses are zero.

4.3 Sensitivity of the Normalized Individual Doses to Variation of Modeling Parameters

Once a set of parameter values is selected for the RMEI and the normalized doses are derived, it is not unreasonable to inquire whether the normalized doses could change substantially if alternative plausible parameter values were used. Often, the derived normalized doses depend heavily on only a few of the calculational parameters. Understanding which of the parameters are important and how the results may change using alternative values is the key to understanding the strengths and limitations of the normalized doses.

4.4 Uncertainties, Variabilities and Sensitivities in the Individual Normalized Doses

This section discusses and quantifies the uncertainties, variabilities, and sensitivities of the normalized doses to the RMEI presented in Table 7-1 (found in Chapter 7 of Volume 1 of the TSD). It would be desirable to evaluate each of the 40 radionuclides listed in Table 7-1. However, the purposes of this section can be accomplished by limiting the evaluation to selected radionuclides which represent each of the important pathways and stages in the life cycle of the recycled scrap metal. For each radionuclide, the limiting normalized dose to the RMEI is associated with a specific stage in the recycling process. Table L.4-1 sorts the radionuclides according to stage and pathway.

Table L.4-1. Limiting Life Cycle Stage and Primary Pathway

Stage	Primary Pathway				
	External Exposure	Inhalation	Ingestion		
			Soot	Food	Ground Water
Scrap metal	Zn-65 Sb-125+D Cs-134 **Cs-137+D***	Ni-59 Ni-63 Tc-99 Mo-93 Ac-227+D	Fe-55		
Finished steel	Mn-54 **Co-60** Ru-106+D Ag-110m+D				
Slag	Mn-54 Nb-94 Ce-144 Eu-152 Ra-226 Ra-228	Pm-147 Th-228+D Th-229+D Th-230 Th-232 Pa-231 U-234 U-235+D **U-238+D** Np-237 Pu-238 Pu-239 Pu-240 Pu-241 Pu-242 Am-241 Cm-244			
Baghouse dust			**Pb-210**		
Airborne effluents				**C-14** **I-129**	
Slag leachate					**Sr-90**
Liquid effluents					
Transportation					
Disposal					

* Radionuclides in bold type were selected for detailed investigation with regard to uncertainty, variability, and sensitivity.

The determination of the limiting life cycle stage and primary pathway for each radionuclide is discussed in Chapters 5 - 7 of Volume 1. Detailed scenario descriptions are found in Appendix H. As discussed in the above-mentioned sections, radionuclides that partition to slag or baghouse dust are reconcentrated, which enhances their potential for exposure. The primary exposure pathways for strong gamma emitters is external exposure, while the nuclides that are primarily beta or alpha emitters deliver doses mainly via the inhalation or ingestion pathways. Individuals residing in the vicinity of steel mills could be exposed to volatile radionuclides in the airborne emissions from the mill. Radionuclides that partition to slag could contaminate ground water in the vicinity of the slag storage facility, as discussed in Section 6.4.1 of the TSD.

The radionuclides listed in bold type in Table L.5-1 were selected for detailed analysis in this section because they are widely present in potential sources of recycled scrap and represent each of the important stages and exposure pathways associated with the recycle of scrap metal from nuclear facilities.

The following sections describe the calculations and assumptions used to derive the normalized doses for the selected key radionuclides: Cs-137+D, U-238+D, Co-60, Pb-210, C-14, and Sr-90+D. By exploring the uncertainties, variabilities, and sensitivities in the normalized doses for these radionuclides, a great deal of insight will be gained into the limitations and strengths of the normalized doses for all the radionuclides.

Each section is organized in the following manner. First the derived normalized doses and risks are stated, and then the values are reproduced using simple hand calculations. The values in Table 7-1 were, in fact, derived using the RECYCLE2 computer program and somewhat more sophisticated models. The use of simple hand calculations in this section accomplishes two objectives. First, it demonstrates the fundamental approach used to derive the values in Table 7-1. Second, by using relatively simple but valid models, insight into the key sources of uncertainty, variability and sensitivity is more readily accomplished. Since simpler models are used, in some cases there are small differences between the values in Table 7-1 and those derived here.

Along with the hand calculations, a discussion is provided of the assumptions used in the analyses and how the results could increase using plausible alternative assumptions. Next, a discussion is provided regarding the doses from other pathways and to other individuals in different stages. The intent of the discussion is to explore the possibility that there may be other pathways or individuals that have the potential for higher normalized doses.

Each section next discusses how the normalized doses could be lower using less conservative assumptions. Finally, each section ends with a listing of the major conclusions.

4.4.1 Cs-137+D

The normalized individual dose for Cs-137+D, as reported in Table 7-1, applies to the scrap cutter and is **8.91 x 10^{-2} mrem/y CEDE per pCi/g of Cs-137 in scrap**. The associated normalized risk from one year of exposure is **6.77 x 10^{-8} lifetime risk of cancer per pCi/g**. The question is, are these reasonable maximum values or are there conditions under which the normalized doses and risks could be substantially higher? Also, are these values unrealistically high? This section attempts to disclose the uncertainty, variability, and sensitivity of these values so that the readers could judge for themselves whether the right balance has been struck in selecting the normalized dose and risk to the RMEI from Cs-137+D.

Stage 1 - Following free release but prior to melting

Once the scrap metal is free-released, it will be shipped to a dealer who may collect, sort, or process the scrap and then ship the scrap to a mill. Whatever the processes, there will be a number of individuals who may be exposed to the scrap from the point of release to when it is sent to the furnace.

The highest dose from gamma-emitting nuclides any individual could receive during this stage is if he or she spent long periods of time in close proximity to large volumes of scrap, thereby receiving the highest possible external exposures. In addition, if the individual were to inhale dust and ingest soot or other material associated with the scrap, he or she would also receive doses from internal exposure. Based on several visits to scrap dealers and mills, we found several individuals that are often exposed in this manner. Specifically, scrap cutters spend most of their working day in close proximity to the scrap and may inhale vapors and ingest soot generated during the cutting of the scrap. The following presents the methods used to derive the normalized doses and risks to these individuals.

External Dose:

The primary pathway from Cs-137+D is external exposure to scrap workers who are assumed to spend 1750 hours per year adjacent to an effectively infinite volume of scrap metal. A simple but reliable method for deriving this dose is to use the dose coefficients listed in Federal Guidance Report No. 12 for external exposure to soil contaminated to an infinite depth. This assumption simply means that the worker spends 1750 hours per year standing on an effectively infinite volume of scrap metal.[2] The normalized dose is derived as follows:

$$D_{Cs-137+D,x} = \frac{c_x \, F_{Ba-137m,x} \, R_{Ba-137m:Cs-137} \, t_e}{f_d}$$

$D_{Cs-137+D,x}$ = normalized annual dose from external exposure to scrap contaminated with Cs-137+D

= 0.0884 mrem/y per pCi/g

c_x = conversion factor from Sv per $Bq \cdot s \cdot m^{-3}$ to mrem per $pCi \cdot hr \cdot g^{-1}$

= 2.13×10^{13}

$F_{ba-137m,x}$ = dose coefficient for external exposure to soil contaminated to an infinite depth with Ba-137m (FGR 12)

= 1.93×10^{-17} Sv per $Bq \cdot s \cdot m^{-3}$

$R_{ba-137m:Cs-137}$ = branching ratio of decay of Cs-137 to Ba-137m

= 0.946

t_e = annual exposure duration

= 1750 hrs/y

f_d = scrap dilution factor (see Section 5.2.1)

= 7.7

In this calculation, it is assumed that, on average, 13% of the scrap processed in the scrap yard is residually contaminated, while the other 87% is clean. Another assumption in the calculation that has an important effect on the results is that the scrap cutter spends 7 hours per day (i.e., 1750 hrs/y) at a distance of 1 meter from an effectively infinite volume of scrap.

At this point, we have an understanding of how the value in Table 7-1 was derived, and two questions come to mind. First, are the modeling assumptions appropriate for this pathway and scenario, and second, are there other pathways and scenarios that could result in a substantively higher normalized dose?

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[2] The applicability to scrap metal of exposure coefficients calculated for soil is discussed in Section 6.3.1 of the main report. A more detailed discussion is found in Appendix H.

With regard to the dilution factor, Appendix G argues that, at most, about 13% of the scrap received by any one processing facility in any given year would be from components that were potentially exposed to radioactivity. If it is plausible for the dilution factor to be 1.0 (i.e., the potentially contaminated scrap is undiluted) at some facility, this normalized dose could be too low by a factor of 7.7. Appendix G argues, however, that a dilution factor of 7.7 is itself highly conservative.[3]

With regard to the exposure times and distances, it could be argued that, in any given year, a cutter may spend more than 1750 hours at his job. For example, if he were to work 10 hours per day at the scrap yard, exclusive of any breaks, the exposure duration would be 2500 hours per year, resulting in a 40% higher normalized dose. The dose rate from an infinite slab source of Cs-137+D is relatively insensitive to the distance over a limited range—reducing the distance would not have any significant effect on this rate.

Assuming a dilution factor of 1 and 2500 hours per year of exposure, the normalized dose from external exposure could be as much as 10 times higher than the value previously calculated. Alternatively, it could also be argued that the amount of dilution has been significantly underestimated given the large throughput of scrap at a scrap yard, and that it is overly conservative to assume that a scrap cutter works full time cutting scrap in close proximity to a virtually infinite volume of scrap. The question is, has the analysis struck the appropriate level of conservatism?

Next, there remains the question whether increased doses via other pathways could substantially increase the normalized dose to the scrap cutter, and are there other exposure scenarios associated with other stages in the life cycle of Cs-137 that could yield higher normalized doses?

Internal Dose:

During the handling and cutting of the scrap, the worker may inhale airborne particles containing Cs-137, or may inadvertently ingest soot containing Cs-137. For the inhalation pathway, the highest average annual concentration of airborne nuisance dust permitted under OSHA regulations is 15 mg/m^3, of which 5 mg/m^3 can be in the respirable range (≤ 5 μm AMAD). In

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[3] Conditions under which the dilution factor could, in theory, approach 1 may occur if economic drivers resulted in the development of regional scrap metal management centers to handle all the potentially contaminated scrap in the region, and these centers established contractual agreements with a limited number of scrap dealers to receive all of the scrap cleared for free release.

order to place an upper end on the possible inhalation dose to the cutter, it can be assumed that the cutter is exposed 1500 hrs/y to 5 mg/m^3 of respirable dust containing 1 pCi/g of Cs-137 (i.e., dust has the same concentration of the Cs-137 as the scrap). Using these bounding assumptions, the inhalation dose would be as follows:

$$D_{Cs-137,h} = \frac{B\ c_i\ f_r\ F_{Cs-137,h}\ t_e\ \chi_d}{f_d}$$

$D_{cs-137,h}$ = 50-year dose commitment from inhalation of Cs-137 during one year (mrem/y EDE per pCi/g in scrap)
= 3.7×10^{-5} mrem/y per pCi/g

B = average breathing rate for an adult worker
= 1.2 (m^3/hr)

c_i = factor for converting from Sv/Bq to mrem/pCi
= 3700

f_r = respirable fraction
= 0.5

$F_{cs-137,h}$ = DCF for inhalation of Cs-137
= 8.63×10^{-9} Sv/Bq (FGR 11)

t_e = 1500 hrs/y

χ_d = concentration of dust in air (dust loading)
= 0.010 mg/m^3
= 0.01 g/m^3

All other terms have the same definitions and values as in the previous equation.

Since the dose from inhalation is three orders of magnitude below the dose from external exposure, changing the exposure parameters for this pathway would not cause a significant change in the calculated normalized dose from Cs-137+D. Hence, the inhalation dose is not a significant contributor to the uncertainty in the calculated normalized dose.

Another possible exposure pathway is the inadvertent ingestion of contaminated soot. As discussed in EPA 89, a high-end estimate of soot ingestion by workers is 480 mg/day. Half this value was used since EPA 89 indicates that the range of values is 0.56 to 480 mg/day, and it would be inappropriate to assume that the maximum daily soot ingestion rate is experienced every day of the working year. Assuming that the soot all contains 1pCi/g Cs-137 , the ingestion dose to the RMEI is estimated as follows:

$$D_{Cs-137,g} = \frac{c_i F_{Cs-137,g} I_s t_e}{f_d}$$

$D_{Cs-137,h}$ = 50-year dose commitment from ingestion of Cs-137 during one year
= 2.9×10^{-4} mrem/y EDE per pCi/g in scrap

$F_{Cs-137,g}$ = DCF for ingestion of Cs-137
= 1.35×10^{-8} Sv/Bq (FGR 11)

I_s = average soot ingestion soot ingestion rate (for a worker in a very dusty environment)
= 30 mg/hr
= .03 g/hr

All other terms have the same definitions and values as in earlier equations.

As the inhalation pathway, the soot ingestion dose is not an important contributor to the normalized dose nor to the uncertainty in the dose from Cs-137.

The total dose calculated above, 0.0887 mrem/y per pCi/g, agrees with the normalized dose for Cs-137 reported in Table 7-1, the slight difference attributable to round-off error.

Other Stages

The above analysis demonstrates that the external exposure pathway dominates for the scrap cutter, and it is unlikely that the normalized dose could be much higher than the derived value for Stage 1 activities. However, the question remains whether there are there other stages in the life cycle of Cs-137 where the normalized dose could be higher.

During the melting of scrap containing Cs-137, most of the cesium volatilizes and becomes part of the baghouse dust. Little if any of the cesium goes to the melt while some (about 5%) partitions to the slag. This raises a series of compound questions. What are the normalized doses associated with Cs-137 in the baghouse dust and slag, and how can we be sure that these normalized doses are less than the normalized dose derived for Stage 1 activities? In addition, what if we are wrong about the fate of the Cs-137? What would be the normalized doses if, for some facilities, under some circumstances, the Cs-137 goes entirely to either the slag or the melt? These questions are explored in this section.

Section 6.2 of the TSD explains that for every 70 tons of scrap metal sent to the furnace, one ton of baghouse dust is generated. This dust will contain 95% of the cesium in the scrap. This means that, if the scrap has an average specific activity 1 pCi/g of Cs-137, the baghouse dust will have an activity of 63 pCi/g. This reconcentration process alerts us that special attention should be given to the baghouse stage of the life cycle when deriving the normalized dose for Cs-137. In fact, on first inspection, it is surprising that the RMEI is a worker in the scrap yard and not one exposed to baghouse dust.

Exposure to baghouse dust is not the limiting scenario for this radionuclide for the following reasons:

1. The mass and the dimensions of the baghouse dust as a source of external exposure are small relative to the effectively infinite size of the scrap in the scrap yard.
2. The individuals spend a small fraction of their time in close proximity to the baghouse dust.
3. Individuals who work inside the baghouse wear some form of respiratory protection.

The question is, are there conditions under which an individual may spend extended periods of time in close proximity to the baghouse dust? Our investigations reveal that this is not likely in the case of the steel mill. However, if such conditions could exist, the normalized dose could be higher than the reported value by several fold, and as high as about 60 fold higher. A 60-fold higher dose could occur if the baghouse dust were allowed to accumulate in large volumes and workers spent virtually full time in close proximity to the dust. A review of the fate of the baghouse dust revealed that it is shipped off site for treatment and ultimate disposal in small incremental batches, and that it is highly unlikely that workers or transporters would spend extended periods of time in close proximity to baghouse dust that was obtained from mills processing scrap from nuclear facilities. However, an investigation of the later stages of processing of baghouse dust after it leaves the steel mill is in progress and will be reported later.

Other stages of the life cycle of Cs-137 (i.e., the slag and the melt) might result in higher normalized doses if it were possible for a large fraction of the Cs-137 to partition to these products. Section E.5.3 of Appendix E gives conclusive evidence that no appreciable amount of cesium can remain in the melt. Under some melting conditions, however, larger fractions of cesium than the 5% assumed in this study could partition to the slag. Table 6.3 shows that radionuclides that partition primarily to the slag have an eight-fold higher concentration in the slag than in the scrap. In the highly unlikely event that 95% of the cesium consistently partitioned to the slag, the normalized dose could be about twice as high as that calculated in the present analysis, all other parameters remaining the same.

Lower End Estimates of the Normalized Individual Doses

Scrap from nuclear facilities could experience much more than an eight-fold dilution. In the U.S., approximately 68 million tons per year of carbon steel is recycled. It is into this flow of scrap metal that the scrap metal from nuclear facilities would enter and be diluted. The total volume of contaminated scrap metal from nuclear facilities is estimated to be about three million tons. Assuming this three million tons is decontaminated and released over a ten-year period (it will probably be closer to 50 years, but some regions of the U.S. may decommission and decontaminate facilities over a ten-year period), the dilution factor will be 68 ÷ 0.3 or ≈230. This is compared to a dilution factor of 7.7 which was used to derive the values in Table 7-1. Hence, additional dilution alone could result in a 30-fold (i.e., 230 ÷ 7.7) reduction in the normalized dose. It is also plausible that the geometry of the contaminated scrap and/or the exposure duration could be smaller, resulting in perhaps a thee-fold reduction. Overall, a 100-fold reduction in the normalized dose is plausible for Cs-137.

Conclusions

It can be concluded that the key assumptions that have the most profound effect on the derived normalized dose for Cs-137, and other radionuclides where the limiting dose is external exposure to scrap, are:

1. The workers experience a 7.7-fold dilution factor.
2. The exposure geometries, distances, and durations greatly reduce the potential for exposure to baghouse dust, and more than offset the enhanced exposure potential due to the reconcentration of Cs-137 in the dust.
3. If these assumptions are incorrect, the normalized dose for Cs-137 could be higher by a factor of about 10 to as high as a factor of 100. Alternatively, using anticipated average dilution factors and reduced occupancy times and geometries, the normalized individual dose could be lower by a factor of 100.

4. Since the object of the analysis is to calculate the dose to the *reasonably maximally* exposed individual in the peak year, it is unlikely that the dose would be significantly higher under *reasonably* anticipated conditions, although it is plausible that the dose to the maximally exposed individual would be lower using a less conservative exposure scenario and a higher dilution factor.

4.4.2 U-238+D

The RMEI for exposure to U-238+D, as reported in Table 7-1, is the slag worker at the mill. His normalized dose in the peak year is calculated to be **0.289 mrem/y CEDE per pCi/g U-238** in scrap, and the associated normalized risk from one year of exposure is **3.55 x 10^{-8} lifetime risk of cancer per pCi/g**. The reason the slag worker is the RMEI for U-238 is that uranium partitions to the slag, where it is reconcentrated. About 8 tons of scrap results in the production of one ton of slag, and 95% of the uranium goes to the slag. As such, the concentration of U-238 in slag is about 8 times that in the scrap. It is therefore not surprising that slag handling constitutes the maximum exposure scenario.

<u>Stage 3 - Exposure to Slag</u>

The slag worker is assumed to spend 1000 hours per year (about half-time) working and standing at the edge of an effectively infinite slab of slag and 1750 hours per year (virtually full-time) in areas where the slag dust loading is 2.6 mg/m^3.

External Exposure:

The following is the method used to derive the upper-end external dose to U-238 to slag workers or to any other worker that spends extended periods of time standing beside large volumes of slag:

$$D_{U-238+D,x} = \frac{c_x \; f_{g,U} \; F_{U-238+D,x} \; t_e}{f_d}$$

$D_{U-238+D,x}$ = normalized annual dose from external exposure to slag contaminated with U-238+D
= 0.0129 mrem/y per pCi/g

$f_{g,U}$ = concentration factor for uranium in slag
= 7.79 (see Table 6-3)

$F_{U-238+D,x}$ = dose coefficient for external exposure to soil contaminated to an infinite depth with U-238 plus its short-lived progeny(FGR 12)
= 7.08×10^{-19} Sv per Bq·s·m^{-3}

t_e = 1000 hrs/y

f_d = 9.1 (see Table 5-1 in Section 5 of the main report)

All other terms have the same definitions and values as in earlier equations.

The result reveals that the external dose is small compared to the total normalized dose of 0.289 mrem/y per pCi/g of U-238 in scrap listed in Table 7-1. As will be demonstrated in the following calculations, the inhalation pathway is the major contributor to the normalized dose for U-238.

Inhalation:

The inhalation dose to slag workers is based on the assumptions that the average annual dust loading is 2.6 mg/m^3, the respirable fraction is 0.51, and the exposure duration is 1750 hours per year. The results of the calculation are as follows:

$$D_{U-238+D,h} = \frac{B \ c_i \ f_{g,U} \ f_r \ F_{U-238,h} \ t_e \ \chi_d}{f_d}$$

$D_{U-238+D,h}$ = 50-year dose commitment from inhalation of U-238+D during one year (mrem/y EDE per pCi/g U-238 in scrap)
= 0.282 mrem/y per pCi/g

f_r = 0.51

$F_{U-238,h}$ = DCF for inhalation of U-238
= 3.20×10^{-5} Sv/Bq (FGR 11)[4]

t_e = 1750 hrs/y

χ_d = 2.6 mg/m^3
= 0.0026 g/m^3

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[4] The inhalation of the short-lived progeny of U-238 in secular equilibrium with the parent makes an insignificant contribution to the internal dose.

All other terms have the same definitions and values as in earlier equations.

This result shows that almost all of the dose to the RMEI from U-238+D listed in Table 7-1 is due to the inhalation pathway.

Several modeling assumptions are important to this calculation. First, a dilution factor of 9-fold is used. As discussed above, this is based on the assumption that only a small fraction of the scrap sent to a mill over the course of a year will be potentially contaminated. The dilution is slightly greater than that for the scrap yard scenario because the supply of scrap metal from the decommissioning of four nearby commercial nuclear power plants would be insufficient to meet the annual demand for scrap of the reference steel mill. Based on the discussion in Appendix G, it is unlikely that the scrap will have less dilution, and it is highly likely that, at most mills, the dilution will be substantively greater.

A slag reconcentration factor of 7.79 is employed. This is based on the assumption that 95% of the uranium in the scrap partitions to the slag and that about 1 ton of slag is produced for every 8 tons of steel. This is not an important source of uncertainty because the partition factor cannot be much larger, and the quantity of slag produced per ton of scrap melted is fairly constant for EAFs. However, if a different melting process were employed—such as vacuum induction melting—the production of slag could be much smaller and the reconcentration factor could be higher, creating the potential for higher exposures. The entire scenario would have to be re-examined in such a case, however, before any conclusions could be drawn about the resulting doses.

The assumed dust loading of 2.6 mg/m^3 and the respirable fraction of 0.51 are based on empirical data. The respirable dust concentration cannot be allowed to exceed 5 mg/m^3 without exceeding OSHA exposure limits. Hence, this value could be as much as 4 times higher. The exposure duration of 1,750 hours per year is close to its upper end value, and an average annual working breathing rate of 1.2 m^3/hr has little uncertainty.

The dust inhalation pathway is based on the assumption that mechanical processes, such as wind erosion and mechanical activity on the slag pile, result in the suspension of particulates containing the radionuclide. Given that the potential exists for particulate suspension, an important modeling assumption is the average outdoor dust loading at the worker location, and the degree to which the dust from the slag may be diluted by uncontaminated sources of dust. In the analysis, it is assumed that all of the dust is from the slag and that the radionuclide concentration in the dust is the same as in the slag.

If the contaminated area were relatively small, it is likely that only a fraction of the dust loading will be from the contaminated material. Given an airborne dust loading of particulates contaminated with U-238, the radiation dose associated with inhalation depends greatly on the particle size distribution of the dust particles containing the radionuclide. Extensive work performed by the International Commission on Radiation Protection (ICRP Publications 30 and 56) reveals that particle sizes greater than 10 microns are generally not respirable. ICRP also indicates that a particle size of about 1 μm can penetrate deeply into the lung. The analysis is based on the assumption that the particles have a 1 μm activity median aerodynamic diameter (AMAD). The aerodynamic diameter of a particle is defined as the diameter of a unit density sphere having the same settling velocity as the particle under consideration. If, in fact, the actual AMAD were greater than 1 μm, the inhalation doses would be substantively lower. For example, if the AMAD were 10 microns, which is still within the respirable range, the dose conversion factor would decrease by about a factor of 2. Alternatively, if the AMAD were substantively smaller than 1 micron (e.g., < 0.1 micron), the dose conversion factor would increase by about a factor of 2.

The possibility also exists that the radionuclide concentration in the dust could be higher or lower than that in the slag. If the radionuclides are located primarily on the smaller particles in the slag, there will be a tendency for the dust to be enhanced in the radionuclide concentration. However, if the radionuclides are primarily on the larger size particles (e.g., > 50 microns) very little of the respirable dust will contain radionuclides (EG&G 84). The possibility of enhancement or discrimination is not addressed in these models but does represent a source of uncertainty. In the case of uranium, EG&G 84 cites a relatively small enhancement factor of 1.5. However, this issue cannot readily be addressed without site-specific empirical data, which, of course, do not currently exist.

Ingestion:

The dose from inadvertent ingestion of particulate matter is derived as follows:

$$D_{U\text{-}238+D,g} = \frac{c_i\, f_{g,U}\, F_{U\text{-}238+D,g}\, I_s\, t_e}{f_d}$$

$D_{U-238+D,g}$ = 50-year dose commitment from ingestion of U-238+D during one year
= 1.7×10^{-3} mrem/y EDE per pCi/g U-238 in scrap

$F_{U-238+D,g}$ = DCF for ingestion of U-238+D
= 1.01×10^{-8} Sv/Bq (FGR 11)

All other terms have the same definitions and values as in earlier equations.

As with the external pathway, the inadvertent ingestion pathway can be disregarded as an important contributor to the normalized dose and to the uncertainty in the normalized dose for U-238.

Other Stages

Because of the reconcentration of the uranium in slag, it is apparent that the other stages could not be limiting. For example, Stage 1 could be associated with exposure settings and durations similar to that of Stage 3, but the worker would not experience the reconcentration effect. Little if any uranium partitions to the melt. Even if more of the uranium went to the melt, it would not undergo the reconcentration that it does in the slag and consequently would produce lower exposures.

About 5% of the uranium ends up in the baghouse dust. However, since such a small amount enters, the uranium concentration in the dust will be about 3 times lower than in the slag. Hence, it is unlikely that Stage 4 could be associated with the limiting dose for uranium. One possible exception to this is if the scrap were contaminated by a uranium compound that became volatilized in the furnace before the charge began to melt. In such a case, a large fraction of the uranium in the scrap could condense in the baghouse dust, and, because of the small mass fraction of the dust, uranium concentrations in the dust could be 8 times higher than in slag. If this were to occur, the inhalation doses could be approximately 8 times higher, assuming comparable atmospheric concentrations of fugitive dust from the furnace as slag dust at the slag handling facility. However, such a scenario is considered unrealistic.

Lower End Estimates of the Normalized Individual Dose for U-238

As discussed above, the normalized dose could be lower by a factor of 30 to account for additional dilution. In addition, the average annual dust loading could easily be 10-fold lower (i.e., 0.26 mg/m^3 as opposed to 2.6 mg/m^3). In addition, the time actually spent in areas containing contaminated dust could be a factor of 2 lower. Overall, the normalized dose for U-238 could be a factor of about 500-fold lower.

Conclusions

The key sources of uncertainty in the normalized individual dose for U-238, and other radionuclides where the primary pathway is inhalation of slag dust, include the scrap dilution factor, the slag reconcentration factor, the dust loading, exposure time, dust particle size distribution, and an enrichment factor. Overall, the normalized individual dose could be conceivably higher by about a factor of 50 and lower by a factor of 500.

4.4.3 Co-60

The normalized RMEI dose for Co-60, as reported in Table 7-1, is **0.899 mrem/y per pCi/g of Co-60 in scrap**, and the associated normalized risk is **6.84 x 10^{-7} lifetime risk of cancer per pCi/g from one year's exposure as reported in Table 7-1.** The limiting pathway is external exposure to the lathe operator.

Stage 2 - The Melt and its Products

Co-60 is chemically similar to iron and tends to stay with the melt. As such, the limiting exposures are associated with the melt and its products. There are a large number of steel products that can be a source of exposure. The lathe operator became the RMEI for Co-60 because a lathe has a massive bed made entirely of cast iron and the operator spends extended periods of time in close proximity to his machine, which tends to maximize his external exposure to Co-60, a very strong gamma-emitter.

Another aspect of this scenario that tends to result in high-end exposures is that *no credit is taken for dilution of the melt with clean scrap.* In theory, a single lathe can be produced from the steel generated by a single melt, all of which can be comprised of potentially contaminated scrap. Unlike the scrap yard, slag handling or baghouse dust maintenance workers, whose exposures depend on the average radionuclide concentrations over the course of a year, the lathe operator could be exposed to a lathe that is made of metal with a scrap dilution factor of 1.

The MicroShield™ computer code was used to calculate the dose to the lathe operator, as opposed to Federal Guidance Report No. 12 methodologies, because the lathe is a finite source. However, for the purpose of performing a qualitative check of the result listed in Table 7-1, the following presents the method used to derive the dose to a person who stands on an infinite slab containing 1 pCi/g of Co-60.

$$D_{Co-60,x} = c_x \, f_{Co,Fe} \, F_{Co-60,x} \, t_e$$

$D_{Co-60,x}$ = normalized annual dose from external exposure to a massive slab of cast iron contaminated with Co-60
= 3.27 mrem/y per pCi/g

$f_{Co-60,Fe}$ = concentration factor of cobalt in cast iron
= 1.01

$F_{Co-60,x}$ = dose coefficient for external exposure to soil contaminated to an infinite depth with Co-60 (FGR 12)
= 8.68×10^{-17} Sv per Bq·s·m^{-3}

All other terms have the same definitions and values as in earlier equations.

This represents an upper-bound estimate of the dose if the source of exposure were effectively an infinite slab. However, because the lathe is of relatively small dimensions, the normalized dose is about 0.9 mrem/y per pCi/g. The implication is that under no reasonable assumptions could the normalized dose be much greater than 3.27 mrem/y per pCi/g, which sets a boundary on the uncertainty in the Co-60 normalized dose. In fact, it could be argued that the probability that any steel or iron component would be made entirely from the melt from undiluted potentially contaminated scrap is small, and this normalized dose may in fact be overly conservative because it does not include a term for dilution.

Other pathways, such as ingestion and inhalation, do not have the potential to contribute to the normalized dose or the uncertainty in the normalized dose for the lathe operator scenario because no appreciable erosion of the machine is likely to occur.

<u>Other Stages</u>

A scrap yard worker is not an RMEI because the scrap in the yard is diluted with uncontaminated metal. However, if a dilution factor of 1 were assumed, the dose to the scrap worker could be comparable to the upper-bound dose to the lathe operator calculated above.

It is not realistic to assume that a significant fraction of the Co-60 could partition to the slag or the baghouse dust to the extent that the normalized dose could be higher than the derived value. For example, even if as much as 10% of the Co-60 partitioned to slag, the reconcentration factor would be about 0.8 (i.e., 0.1 x 8), and the dilution factor would be about 0.11. As such, the Co-60 concentration in the slag would be about 0.08 that in the melt. If one were to assume full time exposure to an effectively infinite slab of slag, the dose would still be a fraction of that derived for the lathe operator.

Lower End Estimates of the Normalized Individual Dose for Co-60

The normalized dose could be lower by as much as a factor of 100 by assuming a greater dilution (a factor of 30), and/or a smaller product and/or shorter exposure times (a factor of 2 to 3).

Conclusions

Because no credit was taken for dilution, the normalized dose for Co-60 is believed to be close to the upper-bound value. In theory, the dose could be 3-fold higher if a steel product far larger than a lathe could be identified. In addition, the normalized dose could be even higher if the person were to spend more than 7 hours per day in close proximity to this product. Overall, it is unlikely that the normalized dose for Co-60, and the other radionuclides where the limiting dose is external exposure to steel products, could be more than 5-fold greater. It is much more likely that the normalized dose would be 100-fold lower.

4.4.4 Pb-210

The normalized dose to the RMEI from Pb-210, as reported in Table 7-1, is **3.08 mrem/y per pCi/g of Pb-210 in scrap, and the associated normalized risk is 4.37 x 10^{-7} lifetime risk of cancer per pCi/g from one year's exposure.** The primary pathway is the ingestion of soot by the furnace operator. Like cesium, lead is volatile and collects in the baghouse dust (i.e., Stage 4), and, since Pb-210 is a beta emitter, the principal pathways of exposure are inhalation and ingestion.

Stage 4 - Baghouse Dust

The soot ingestion dose for Pb-210 is derived using the same equation described above for slag, except it is assumed that the furnace operator is exposed to fugitive dust and soot that escape capture by the emissions control system, and thus are assumed to have the same composition as the baghouse dust. The dose is calculated as follows:

$$D_{Pb-210+D,g} = \frac{c_i \, f_{d,Pb} \, F_{Pb-210+D,g} \, I_s \, t_e}{f_d}$$

$D_{Pb-210+D}$ = 50-year dose commitment from ingestion of Pb-210+D during one year
= 2.65 mrem/y EDE per pCi/g Pb-210 in scrap

$f_{d,Pb}$ = concentration factor of lead in baghouse dust
= 63.3

$F_{pb-210,g}$ = DCF for ingestion of Pb-210+D
= 1.96×10^{-6} Sv/Bq (FGR 11)

All other terms have the same definitions and values as in earlier equations.

The remainder of the normalized dose is from the inhalation pathway:

$$D_{Pb-210+D,h} = \frac{B \, c_i \, f_{d,Pb} \, f_r \, F_{Pb-210+D,h} \, t_e \, \chi_d}{f_d}$$

$D_{pb-210+D,h}$ = 50-year dose commitment from inhalation of Pb-210+D during one year
= 0.417 mrem per pCi/g Pb-210 in scrap

f_r = 0.58

$F_{pb-210+D,h}$ = DCF for inhalation of Pb-210+D
= 6.04×10^{-6} Sv/Bq (FGR 11)[5]

χ_d = 2.2 mg/m^3
= 0.022 g/m^3

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[5] The inhalation of the short-lived progeny of U-238 in secular equilibrium with the parent makes an insignificant contribution to the internal dose.

The sum of the two pathways is 3.07 mrem/y per pCi/g of Pb-210 in scrap, which agrees well with the value in Table 7-1, the difference being attributable to round-off error. There is also a contribution from external exposure to Pb-210 in the furnace. Because of the low energy, weak intensity of the Pb-210 γ-ray and the shielding afforded by the walls of the furnace, this contribution is ten orders of magnitude less than the ingestion dose and is therefore neglected in the calculation presented above.

These calculations utilize a number of conservative assumptions that result in a high end estimate of the normalized dose, including:

1. The assumed 1 micron particle size distribution of the dust results in a high-end normalized inhalation dose.
2. The soot ingestion rate is a high-end value
3. All ingested soot is assumed to originate in the furnace emissions

Other Stages

Given the fact that Pb-210 is a beta emitter and is primarily a source of internal exposure, it is unlikely that other pathways and stages could result in higher exposures than that of the furnace operator. Specifically, the analysis uses a 63-fold reconcentration, a high dust loading, and exposure time corresponding to a full-time assignment with no rotation of tasks.

Lower End Estimates of the Normalized Individual Dose for Pb-210

The normalized dose could be lower by as much as a factor of 500 by assuming a greater dilution (a factor of 30), a 10-fold lower dust loading, and/or shorter exposure times (a factor of 2 to 3).

Conclusions

The doses could be higher under the following conditions:

1. The dilution factor equals 1.
2. The quantity of dust produced per ton of scrap processed is lower than assumed, resulting in a higher level of reconcentration. The value used is based on EPA-derived emission factors and represents an industry average. The actual quantity of dust varies from one mill to another and from one batch of scrap metal to another.

3. The time-weighted average of respirable dust is higher than 1.3 mg/m^3. The dust loading could, in theory, be as high as the OSHA limit of 5 mg/m^3.

4. The exposure duration could be greater than 1,750 hours per year if the operator spends the entire 8-hour day in the vicinity of the furnace or if he works overtime. However, the increase would be relatively small.

5. The average daily soot ingestion rate is higher than 30 mg/hr (240 mg/d). The high end value reported in EPA 89 is 480 mg/d. As such, the soot ingestion rate could be twice the value used. However, the likelihood that the high end daily soot ingestion rate would persist for 250 days per working year seems unlikely.

Overall, the normalized dose could, theoretically, be up to 20-fold higher and 500-fold lower. The lower value would be due to greater dilution (a factor of 30), lower dust loading (a factor of 10), and reduced occupancy times (a factor of 2).

4.4.5 C-14

The normalized RMEI dose for C-14, as reported in Table 7-1, is 8.66×10^{-4} mrem/y per pCi/g of C-14 in scrap, and the associated normalized risk is 4.28×10^{-10} lifetime risk of cancer from one year's exposure per pCi/g of scrap. The limiting pathway is the consumption of foods grown offsite which are contaminated by airborne emission from the mill.

C-14 and I-129 are unique in that they volatilize during the melt, escape the baghouse and are released to the atmosphere. As such, they have the potential to cause offsite exposures (Stage 5).

Stage 5 - Offsite Exposure from Airborne C-14 Emissions from the Mill

The normalized dose from C-14 is derived on the assumption that all the C-14 in the melt becomes airborne, is discharged to the atmosphere, is transported offsite, and exposes individuals who are assumed to live 1 km downwind of the site. These individuals are exposed by a number of pathways (e.g., inhalation, external exposure, food ingestion). However, as will be demonstrated, the limiting pathway is food ingestion because C-14 is taken up by plants and is a pure beta emitter (i.e., the external exposure pathway is insignificant).

The ingestion doses for vegetables, milk, and beef are derived by first determining the annual atmospheric release rate of C-14:

$$Q_{C\text{-}14} = \frac{f_{C,v}\, T_s}{f_d}$$

$Q_{C\text{-}14}$ = annual average atmospheric release rate of C-14
= 346 pCi/s per pCi/g C-14 in scrap
$f_{C,v}$ = fraction of carbon volatilized from the scrap during melting
= 0.73
T_s = annual average throughput of scrap melted by steel mill
= 150,000 tons/y
= 4,312 g/s

This release is dispersed in the atmosphere, and the concentration of C-14 in the air at the off-site receptor location is obtained by multiplying the release rate by the atmospheric dispersion factor, which is calculated by EPA's CAP-88 computer code, used for assessing compliance with the Clean Air Act. More details are presented in Section 6 of the main report and in Appendix H.

The concentrations of C-14 in the vegetables, milk, and beef obtained at the receptor location are derived using the specific activity approach. This approach is based on the fact that the carbon in all organisms is ultimately obtained by photosynthesis from CO_2 in the air. As such, the average annual specific activity of C-14 in the atmosphere, expressed in units of pCi of C-14 per gram of carbon in $C0_2$ in the atmosphere, is also the specific activity of C-14 in all living organisms that obtain their food at that location. The primary pathway is the consumption of vegetables. We first calculate the specific activity of C-14 in vegetables grown at the receptor site:

$$C_{C\text{-}14,v} = \frac{C_{C,v}\, Q_{C\text{-}14}\, (\chi/Q)}{C_{C,a}}$$

$C_{C\text{-}14,v}$ = concentration of C-14 in vegetables
= 2.3×10^{-3} pCi/g
$C_{C,v}$ = concentration of carbon in vegetables (w/w)
= 0.11

χ/Q = annual average atmospheric dispersion factor for maximum downwind sector, as calculated by CAP-88 (see Section 6 of the main report and Appendix H)
= 9.67×10^{-6} s/m^3
$C_{C,a}$ = concentration of carbon in the atmosphere (in CO_2)
= 0.16 g/m^3

The annual dose from the consumption of vegetables is calculated as follows:

$$D_{C-14,v} = c_i C_{C-14,v} F_{C-14,g} I_v$$

$D_{C-14,v}$ = 50-year dose commitment from ingestion of C-14 in vegetables during one year
= 5.86×10^{-4} mrem/y EDE per pCi/g C-14 in scrap

$F_{C-14,g}$ = DCF for ingestion of C-14
= 5.64×10^{-10} Sv/Bq (FGR 11)
I_v = vegetable consumption rate
= 1.22×10^5 g/y

The remainder of the dose is from the consumption of other foods (e.g., milk, meat, etc.)

Inspection of this calculation reveals that the doses could be higher if less scrap dilution were assumed, the distance to the receptor were less than 1 km, resulting in a higher atmospheric dispersion factor, and the individual consumed more locally grown vegetables than assumed.

The maximum annual dose, given the C-14 release rate and atmospheric dispersion cited above, would result if all the carbon in the individual's tissues came to equilibrium with the carbon in the atmosphere, i.e., the individual obtained all his or her nutrients from food produced in the immediate vicinity, over a long enough period that all the carbon in his or her tissues was replaced. The C-14 concentration in the individual's tissues would then be:

$$C_{C-14,t} = \frac{C_{C,t} Q_{C-14} (\chi/Q)}{C_{C,a}}$$

$C_{C\text{-}14,t}$ = concentration of C-14 in tissue
= 4.8×10^{-3} pCi/g
$C_{C,v}$ = concentration of carbon in human tissue(w/w)
= 0.23

The dose from such a tissue concentration of C-14 can be calculated as follows:

$$D_{C-14,t} = C_{C-14,t}\, t_e\, \Delta$$

$D_{C\text{-}14,t}$ = annual dose from equilibrium concentration of C-14 in tissue
= 4.43×10^{-3} mrem/y
t_e = 8776 hrs/y
Δ = absorbed dose rate from radioactive decay of C-14
= 0.105 g·rad/μCi·hr
= 1.05×10^{-4} g·mrad/pCi·hr

Hence, this represents an upper bound estimate of the C-14 dose given the source term and atmospheric dispersion factor.

Other Stages

C-14 can be entirely retained in the melt, depending on the melting practice. In such a case, the RMEI would be the scrap cutter. His normalized dose would be 1.5×10^{-5} mrem/y per pCi/g, over 50-fold less than that of the off-site resident.

Lower End Estimates of the Normalized Individual Dose for C-14

The dose from C-14 could be 100-fold lower assuming increased dilution (a factor of about 30), further distance of the farm from the mill (a factor of about 2), and a smaller fraction of his food is obtained locally (a factor of about 2). In addition, C-14 may not be volatile, depending on the melting practice. Under such conditions, the RMEI would be the scrap cutter.

Conclusions

The key sources of uncertainty in the normalized dose for C-14 are the scrap dilution factor, the atmospheric transport factor (which depends primarily on the distance to the nearest downwind receptor), and the quantity of food obtained locally by the receptor. In theory, the normalized dose for C-14 could be 50-fold higher based on (1) the elimination of dilution (factor of 8), (2) the possibility that the nearby farm is closer to the mill than assumed in the base case (a factor of about 2), and (3) the assumption that the farmer obtains a larger fraction of his food from his farm (a factor of about 3). Theoretically, the normalized dose for C-14 could be lower by up to 4 orders of magnitude if the C-14 remained in the melt.

4.4.6 Sr-90

The normalized RMEI dose for Sr-90, as reported in Table 7-1, is **3.03 mrem/y per pCi/g of scrap**, and the associated normalized risk is **5.15 x 10^{-7} lifetime risk of cancer from one year of exposure**. The limiting pathway is the ingestion of ground water contaminated from leachate from slag stored outdoors at the mill.

Ground water contamination (Stage 6) is the limiting pathway for Sr-90 for a number of reasons. Sr-90 preferentially partitions to slag and therefore is reconcentrated in slag by a factor of about 8. In addition, since Sr-90 and its progeny are pure beta emitters, the only significant dose would be from internal exposure. Finally, strontium has a relatively high potential to leach from the slag, infiltrate through the soil and contaminate an underlying aquifer.

The concentration of Sr-90 in the aquifer is calculated by the following equation, which combines Equation 6-20 in Chapter 6 of Volume 1 with the expression for the ground water dilution factor that immediately follows it:

$$C_{Sr-90,w} = \frac{D\, f_{Sr,g}\, F_{Sr}\, l\, \rho_g\, e^{-\lambda_{Sr-90} \Delta t_{Sr}}}{d_w\, f_d\, K_s\, i}$$

$C_{Sr-90,w}$ = normalized concentration of Sr-90 in aquifer
= 0.0273 pCi/mL per pCi/g Sr-90 in scrap
D = depth of slag layer
= 100 cm
$f_{Sr,g}$ = concentration factor of strontium in slag
= 7.79
F_{Sr} = fraction of strontium leached from slag in one year
= 0.336
l = length of slag pile
= 94 m

ρ_g = specific gravity of slag
= 2
$\lambda_{Sr\text{-}90}$ = radioactive decay constant of Sr-90
$\Delta t_{Sr\text{-}90}$ = travel time of strontium to the aquifer
= 75.23 y (see Section 6.4.1 of main report)
d_w = screened depth of well
= 3 m
K_s = saturated hydraulic conductivity of aquifer
= 5.5×10^5 cm/y
i = hydraulic gradient
= 0.02

A detailed discussion of these parameters can be found in Section 6.4.1 of Volume 1 of the TSD. The main uncertainties in this calculation are the fraction of strontium leached from slag, which is based on the upper-bound value of the diffusion coefficient as listed in a preliminary report on experiments on leaching of EAF slag recently conducted at the Brookhaven National Laboratory (BNL). This value obviously cannot exceed 1. It is not likely that the value can be significantly higher than that used in the analysis, unless the slag is divided into much finer particles than the 1 cm pebbles assumed here. The dimensions of the slag pile are based on one year's production at the reference facility, and are not likely to change significantly.

As stated before, the above equation is the product of the expressions for the Sr-90 concentration in the pore water in the soil, the dilution in the aquifer and the radioactive decay during the transit time to the aquifer. The aquifer dilution factor is calculated to be 0.142 (see Section 6.4.1). This factor obviously cannot be greater than 1. A dilution factor of 1 implies that all of the water entering the well originated in rainwater that infiltrated the slag pile. Given the model employed, this would be the case if the volumetric flow rate of the aquifer were less than about 16 m/y, instead of the 110 m/y currently modeled. The dilution factor is based on the assumption that the well is immediately downgradient of the slag pile. If the well is situated at a distance, the dilution factor could be many times greater.

The last term in the equation, $e^{-\lambda\Delta t}$, which translates to a 0.166 multiplier, accounts for the decay of the Sr-90 in transit to the underlying aquifer. The transit time in the 4 m unsaturated zone is estimated to be about 75 years, as discussed in Section 6.4.1 of the TSD. The critical values are the depth of the aquifer and the distribution coefficient (K_d). A shallow aquifer—such as the 4 m depth used in the analysis—and the low-end K_d value of 15 cm^3/g tend to minimize the travel time and thus maximize the Sr-90 concentration in the drinking water. Substantially increasing the K_d or the depth of the aquifer would result in the Sr-90 essentially decaying away before reaching the aquifer.

The K_d is an empirically derived constant that relates the radionuclide concentration in soil to that in water that saturates the soil. A review of the K_d values for strontium summarized in EPA 94 show a range of 0.01 to 32,000. This parameter is the single largest contributor to the uncertainty in the normalized dose for Sr-90. The selection of 15 is somewhat arbitrary as applied to the soil beneath the slag. In fact, since the slag creates an alkaline environment, the K_d is likely to be higher than 15. If strontium had a K_d of 110, which is assigned to iron and niobium in Table 6-5, the travel time through the same 4 m of soil would increase to 545 y. During such a travel time, the activity of Sr-90, with a half-life of 29 y, would be reduced by an additional five orders of magnitude, with a corresponding effect on the calculated dose. In such a case, the RMEI for Sr-90 would be the slag yard worker.

The dose to the RMEI is calculated as follows:

$$D_{Sr-90+D,g} = c_i \, C_{Sr-90,w} \, F_{Sr-90+D,g} \, I_w$$

$D_{Sr-90+D}$ = 50-year dose commitment from ingestion of Sr-90+D in drinking water during one year
= 1.60 mrem/y EDE per pCi/g Sr-90 in scrap
$F_{sr-90,g}$ = DCF for ingestion of Sr-90+D
= 4.14×10^{-8} Sv/Bq (FGR 11)
I_w = drinking water consumption rate
= 7.3×10^{5} mL/y

All other terms have the same definitions and values as in earlier equations.

The result is consistent with the value reported in Table 7-1. Given the calculated value of the Sr-90 concentration, the dose is an upper-end value. If the individual consumed less than 2 liters of well-water per day, his or her dose would be correspondingly less.

Other Stages

If the ground water pathway were eliminated, the limiting dose from Sr-90 would be to slag workers inhaling and ingesting slag particles (Stage 3). The normalized dose would be 8.7×10^{-3}

mrem/y, as listed in Appendix K, which is over two orders of magnitude less than that associated with the ground water pathway. The uncertainties associated with these pathways are discussed above.

Conclusions

The uncertainties associated with the normalized dose for Sr-90 are very large. However, since we assumed a relatively low K_d, especially knowing that the slag has a high pH, which tends to dramatically increase the Kd, it is likely that the ground water pathway will be eliminated once we obtain better information on the K_d. That being the case, the normalized dose will likely be reduced by about a factor of about 175. This new normalized dose will have the same types of uncertainties described above for the inhalation and ingestion pathways for slag.

4.5 Summary of Key Sources of Uncertainty in the Individual Normalized Doses

Table L.4-2, along with Figure L.4-1, summarize the results of the uncertainty analysis. As may be noted, Table L.4-2 groups the radionuclides by the stage in the life cycle of the radionuclide during the recycling process and by exposure pathway. This grouping serves our purposes because the controlling assumptions and parameters differ as a function of the life cycle stage and pathway. For each grouping of radionuclides, an upper end multiplier and a lower end divisor is assigned. These are the multipliers and divisors that should be used to bound the values of the normalized doses presented in Table 7-1 of the TSD. For example, in Table 7-1, the normalized individual dose for Co-60 is 0.899 mrem/yr per pCi/g of Co-60 in released scrap metal. According to Table L.4-2, this normalized dose could be as much as 5 times larger or 100 times smaller using plausible alternative assumptions. The right hand column of Table L.4-2 summarizes the bases for the multipliers. Figure L.4-1 applies the multipliers and divisors to each of the normalized doses and placed the values on a bar chart.

The multipliers and divisors are largely based on professional judgment and are designed to provide an order of magnitude estimate of the uncertainties and variabilities in the normalized doses. In Figure L.4-1, the upper end values should be thought of as the highest possible normalized doses for the members that make up the limiting population group for a given radionuclide. They could also be thought of as the highest possible value for the normalized dose for the RMEI for each radionuclide. The lower end values could be thought of as a typical value for the normalized doses for the members of the limiting population group, or as the lowest possible value for the normalized doses for the RMEI. The next step in the process is to use this information to determine the scope and approach for a more quantitative analysis of uncertainties, perhaps using Monte Carlo techniques.

Table L.4-2. Uncertainty/Variability in Normalized Individual Doses

Radionuclides	Limiting Stage	Primary Pathway	Upper End Multiplier	Lower End Divisor	Bases
Zn-65* Sb-125 Cs-134* Cs-137*	Scrap yard	External exposure	10	100	Upper end due to eliminating dilution factor.
					Lower end due to additional dilution (30 fold), reduced occupancy and increased distance (3).
Ni-59 Ni-63 Mo-93 Tc-99 Ac-227+D		Inhalation	10	500	Upper end due to eliminating dilution factor.
					Lower end due to additional dilution (30 fold), reduced occupancy (2), and reduced dust loading (10)
Fe-55		Soot ingestion	10	500	Upper end due to eliminating dilution factor.
					Lower end due to additional dilution (30), reduced occupancy (2), and reduced soot ingestion (10)
Mn-54 Co-60 Ru-106 Ag-110m+D	Metal products	External exposure	5	100	Upper end due to increase in size of component and occupancy time (5).
					Lower end due to application of a dilution factor (30) and lower occupancy time and smaller size component (3).
Nb-94 Ce-144+D Eu-152 Ra-226+D Ra-228+D Th-228+D	Slag pile	External exposure	40	100	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (4).
					Lower end due to additional dilution (30) and smaller contaminated area and occupancy time (3).
Pm-147 Th-229/230/232 Pa-231 U-234/235/238 Np-237 Pu-all Am-241 Cm-244	Slag pile	Inhalation	20	500	Upper end due to elimination of dilution factor (9) and increased occupancy time and slag partition (2).
					Lower end due to additional dilution (30), lower dust loading (10), and lower occupancy time (2).
Pb-210	Mill	Ingestion	20	500	Upper end due to elimination of dilution factor (8) and increased occupancy time and slag partition (2).
					Lower end due to additional dilution (30), lower soot ingestion (10), and lower occupancy time (2).
C-14	Offsite exposure to airborne emissions	Ingestion of food	50	NA†	Upper end due to elimination of dilution factor (8), closer location (3), increased intake of crops (2).
I-129				100	Lower end due to additional dilution (30), further distance (2), less intake (2).
Sr-90	Ground water contaminated by slag leachate	Ground water ingestion	50	NA†	Upper end due to less dilution in ground water.
					Lower end due to elimination of ground water due to increased transit time, and soot ingestion becomes the limiting pathway.

* These radionuclides partition to baghouse dust. If it is plausible for individuals to be exposed to reconcentrated stages of the metal recovery process for prolonged periods of time, the upper end multiplier for these radionuclides could be as high as a factor of 100.

† A lower limit for these pathways in not applicable, since the lowest limiting dose will be due to a different pathway (see text).

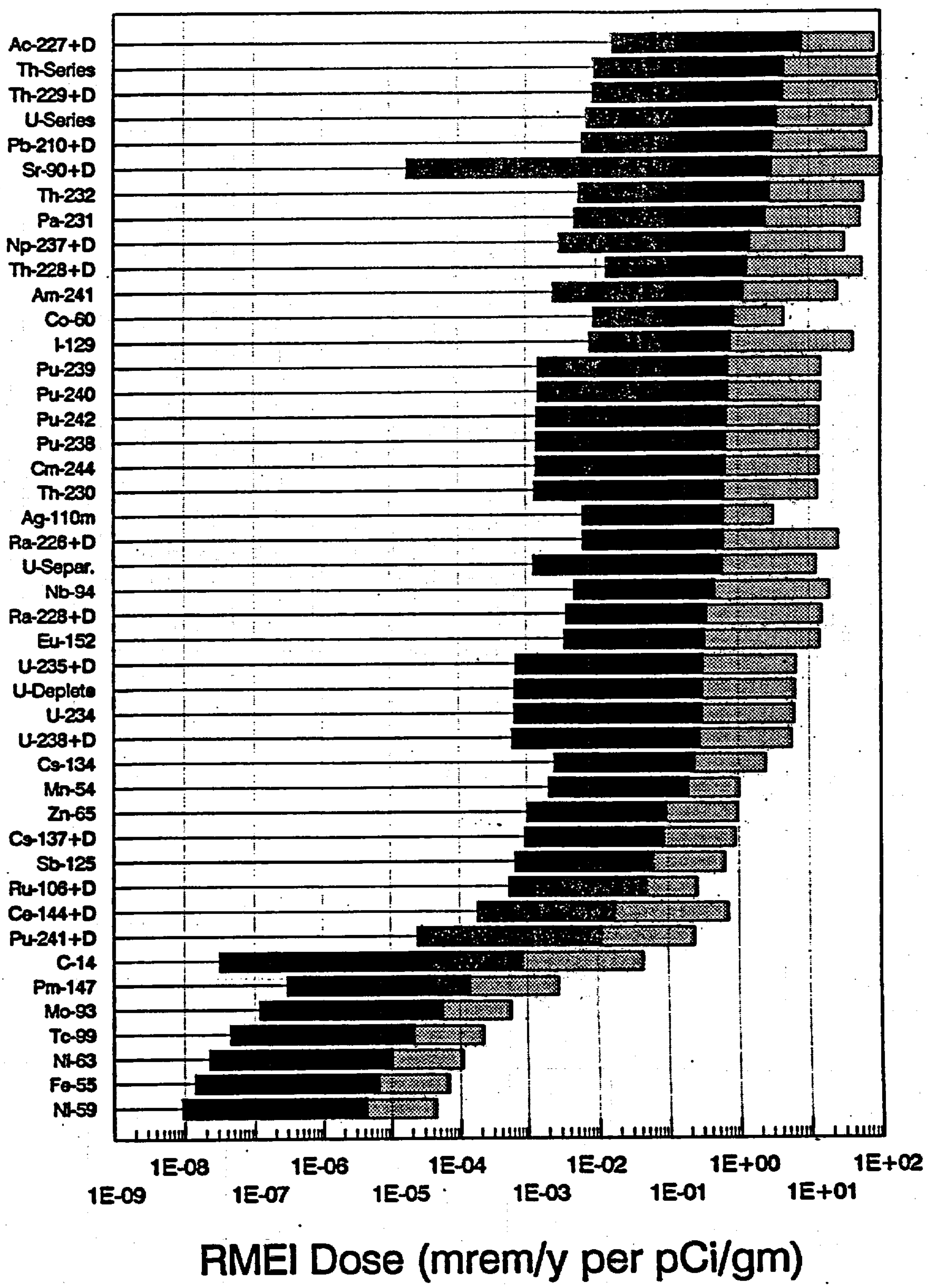


Figure L.4-1. RMEI Dose (mrem/y per pCi/gm)

4.5.1. Stage 1 - Scrap Metal Before Melting

Before the scrap is melted, an individual handling the scrap can be exposed to external radiation, the inhalation of suspended particles coming off the scrap or while the scrap is being cut up, and by the ingestion of contaminated soot. Based on the results of our models, this stage in the life cycle of the scrap is limiting for ten of the 40 radionuclides analyzed.

External Exposure. Of the ten radionuclides, external exposure is the limiting pathway for Zn-65, Sb-125, Cs-134 and Cs-137. The upper end multiplier is 10 and the lower end divisor is 100.

Upper End Multiplier

Inspection of the external exposure model reveals that it was assumed that the individual spends seven hours per day about one meter from an effectively infinite slab of scrap metal. It was also assumed that only 13% (i.e., an 8-fold dilution) of the scrap handled during a given year is contaminated, while the rest is clean. This assumption is based on the proportion of all scrap metal generated by the decommissioning and decontamination of four commercial nuclear plants. It could be argued that, in a given year, a worker could put in some overtime, and the fraction of the contaminated metal handled by the scrap yard is greater than 13%, perhaps close to 100%. On this basis, an upper end multiplier of 10 is selected.

Lower End Divisor

The released scrap from a nuclear facility could experience much more than an eight-fold dilution. In the U.S., approximately 68 million tons per year of carbon steel is recycled. It is into this flow of scrap metal that the scrap metal from nuclear facilities will enter and be diluted. The total volume of contaminated scrap metal from nuclear facilities is estimated to be about three million tons. Assuming this three million tons is decontaminated and released over a ten-year period (it will probably be closer to 50 years, but some regions of the U.S. may decommission and decontaminate facilities over a ten-year period), the dilution factor will be $0.3 \div 68$ or 4.4×10^{-3}. This is as compared to a dilution factor of 0.13 which was used in the base case. Hence, additional dilution alone could result in a 30-fold (i.e., $0.13 \div 4.4E\text{-}3$) reduction in the normalized dose. It is also plausible that the geometry of the contaminated scrap could be smaller than an effectively infinite slab, resulting in perhaps a three-fold reduction. Overall, a 100-fold reduction in the normalized dose is plausible.

Inhalation. For five of the Stage 1 radionuclides (Ni-59, Ni-63, Mo-93, Tc-99 and Ac-227+D), the primary pathway is inhalation because they are primarily alpha, beta or low-energy x-ray emitters.

Upper End Multiplier

The normalized doses for the five Stage 1 radionuclides that are limited by the inhalation pathway could be 8-fold higher due to the elimination of the 0.13 dilution factor. Although, in theory, the concentration of dust could be somewhat higher, this increase would be marginal, since an average annual value of 10 mg/m^3 was assumed which is comparable to the OSHA PEL of 15 mg/m^3. Again, the worker could spend more than the assumed six hours per day inhaling the dust. As such, a 10-fold upper estimate is selected.

Lower End Divisor

The lower end divisor includes the factor of 30 to account for additional dilution. In addition, the average annual dust loading could easily be 10-fold lower (i.e., 1 mg/m^3 as opposed to 10 mg/m^3). In addition, the time actually spent cutting the contaminated scrap with a torch might be only ½ the overall time devoted to this task. Overall, a 500-fold divisor is selected.

Soot Ingestion. The normalized dose for one radionuclide, Fe-55, is dominated by soot ingestion. The upper end and lower end uncertainties for ingestion of Fe-55 are approximately 10 and 500, as they are for inhalation—the comparable pathway for this nuclide. Although the soot ingestion rate could be twice as high as the one assumed, this pathway accounts for just 53% of the dose, the remainder being due to inhalation of dust. ~~Thus,~~ an upper end multiplier of 10 is an appropriate order-of-magnitude estimate for this nuclide. Similarly, the lower end divisor includes a factor of about 30 for additional dilution, a factor of 10 for reduced soot ingestion, and a factor of 2 for reduced time spent generating the contaminated soot.

4.5.2. Stage 2 - Melt and Melt Products

The normalized individual doses for four of the 40 radionuclides are limited by external exposure to steel products made from scrap. Since these products do not include a dilution factor, the doses could be higher only if the product were much larger than the lathe and/or more time were spent in close proximity to the product. This combination of assumptions could theoretically increase the normalized dose by much as a factor of five.

The normalized dose could be lower by as much as a factor of 100 by assuming a greater dilution, a smaller product and/or shorter exposure times.

4.5.3. Stage 3 - Slag and Slag Uses

The normalized individual doses for 22 of the 40 radionuclides are limited by exposure to slag stored at the steel mill. Slag constitutes the limiting pathway because these radionuclides reconcentrate about 8-fold in slag.

External Exposure. The normalized doses for five of these 22 radionuclides (Nb-94, Ce-144+D, Eu-152, Ra-226+D and Ra-228+D) are primarily from external exposures because these nuclides are strong gamma emitters. The upper end values could be higher by about a factor of 40 due to (1) elimination of the dilution factor, (2) a smaller volumes of slag per batch of process steel (thereby resulting in a higher reconcentration factor), and (3) an increased source size and duration of exposure (together a factor of no more than 4).

Inhalation. The normalized dose for the remaining 16 of these 22 radionuclides are limited by inhalation of slag dust. Inhalation is limiting because these radionuclides are primarily alpha or beta emitters. The upper end multiplier for these radionuclides is estimated to be about 20 primarily due to the elimination of the dilution factor and, to a lesser degree, due to increased dust loading. The lower end divisor is 500, for reasons discussed above.

4.5.4. Stage 4 - Mill Operations Baghouse Dust

The normalized dose for only one radionuclide, Pb-210, is limited by the ingestion of soot from EAF baghouse dust. This occurs because Pb-210 partitions to the baghouse dust and is a beta emitter. The upper end multiplier of 20 and lower end divisor of 500 were selected for the same reasons discussed above for soot ingestion.

4.5.5. Stage 5 - Offsite Contamination from Airborne Emissions

Only two of the 40 radionuclides (I-129 and C-14) are volatile and can escape the baghouse. These radionuclides can cause offsite exposure by contaminating locally grown foods. The upper end multiplier of 50 is based on (1) the elimination of dilution (factor of 8), (2) the possibility that the nearby farm is closer to the mill than assumed in the base case (a factor of about 2), and (3) the assumption that the farmer obtains a larger fraction of his food from his farm (a factor of about 3).

The lower end divisor for I-129 is 100 is due to increased dilution (a factor of about 30), assumed further distance of the farm from the mill (a factor of about 2), and a smaller fraction of his food is obtained locally (a factor of about 2). C-14 may not be volatile, depending on the melting practice. Under such conditions, the RMEI would be the scrap cutter.

4.5.6. Stage 6 - Ground Water Contamination from Slag Leachate

The normalized dose for one radionuclide, Sr-90, is limiting due to ground water contamination by slag leachate. This occurs because of a combination of three characteristics of Sr-90: it partitions to slag, it is a pure beta emitter, and it has a low binding ability to soil. The high end multiplier for the Sr-90 normalized dose is 50 based on the assumption that the transit time to the receptor could be reduced by assuming a lower retardation factor and a shallower aquifer, as well as no dilution of the potentially contaminated scrap metal that produced the slag.

The lower end limiting dose would be the same as it is for inhalation of dust from the slag pile, since, if less conservative ground water transport assumptions were used, the ground water pathway would no longer be limiting, and the slag dust inhalation pathway would dominate.

4.5.7. Special Cases

Several radionuclides partition to baghouse dust and are reconcentrated in the dust approximately 60-fold. However, exposure to baghouse dust is not limiting for these radionuclides. Specifically, Cs-137, Cs-134, and Zn-65 partition to the baghouse dust but external exposure to scrap is the limiting scenario and pathway, because it is believed that, while the dust is at the mill, no workers are in close proximity to the dust for prolonged periods of time, as they are with the scrap metal. In addition, after the dust leaves the mill, it is shipped to processing facilities where the dust is diluted with large volumes of baghouse dust from other mills such that the potential for exposures is markedly reduced. However, the processing of the dust and the recovery of commercially valuable metals may cause a reconcentration of these nuclides. Zn-65 will concentrate in the zinc that is recovered from the dust, while Cs-134 and Cs-137 will be reconcentrated in the waste product of the recovery process. If workers were to be exposed to large quantities of these reconcentrated stages of the process, the potential exists for significantly higher normalized doses for these radionuclides. Specifically, referring to Table L.4-2, the upper end multiplier is assigned a value of 10 for Cs-134, Cs-137, and Zn-65. If exposure scenarios exist where individuals can come into close contact with such reconcentrated stages for prolonged periods of time, the upper end multiplier could be a factor of 100, instead of 10.

5. UNCERTAINTY IN THE NORMALIZED COLLECTIVE DOSES AND RISKS

Table 9-15 in Volume 1 of the TSD presents the normalized collective doses for selected radionuclides. The doses are reported in units of person rem per Ci of each radionuclide in free released scrap metal. A full description of the methods used to derive the normalized collective doses and risks is provided in Chapter 9.

This section discusses the uncertainty in the values presented in Table 9-15. In general, the results reveal that the collective impacts could be only marginally higher, less than a factor of 3. However, the values could be lower by perhaps an order of magnitude.

As described in Chapter 9, the normalized collective impacts were derived by using a two-step process. First, an estimate was made of the collective impacts per unit activity released (all results are expressed in terms of impacts per Curie released) as if the Ci ended up entirely in a given product or byproduct. This is referred to as the "unweighted normalized collective dose." Next, an estimate is made of how the radioactivity that is released may partition among the products or byproducts associated with recycling. By multiplying the unweighted normalized collective dose by the partitioning fractions, and then summing the results, an estimate is made of the weighted normalized collective doses, which are the values presented in Table 9-15.

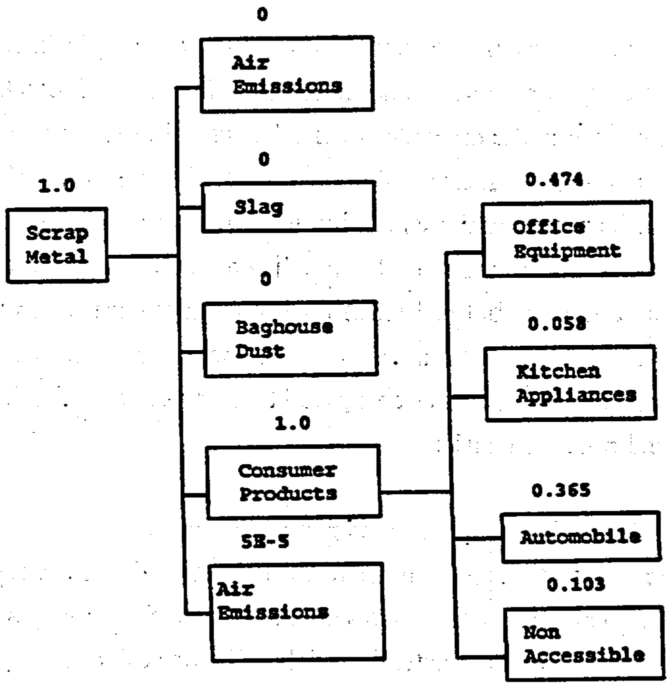
In the section on individual normalized doses, the concept of variability was emphasized. The reason was that the calculation was concerned with the doses to individuals, which can be highly variable among individuals. In this section, the opposite is true. This section is concerned with the time-integrated collective impacts for each radionuclide on a national scale. As such, there is no variability, only uncertainty in a future, potentially real, but unknown value. Issues related to variability from site to site or person to person are not relevant. There is only uncertainty in determining the correct value for an unknown but real value for each parameter used in the calculation. This is an important concept because it means that the "uncertainty" in the results is likely to be much smaller for the collective impacts as compared to the uncertainty/variability in the derived values for the individual doses and risks.

This section is divided into subsections addressing selected radionuclides. Each subsection is organized in the following manner. First the derived normalized collective doses are reproduced using simple hand calculations. The values in Table 9-15 were, in fact, derived using a computer program and somewhat more sophisticated mathematical models. The use of simple hand calculations in this section accomplishes two objectives. First, it clearly discloses how the collective doses and risks were derived. Second, by using relatively simple but valid models, insight into the key sources of uncertainty and sensitivity is more readily accomplished. Since simpler models are used, in some cases there are small differences between the values in Table 9-15 and those derived here.

Along with the hand calculations, a discussion is provided of the assumption used in the analyses and how the results could change using plausible alternative assumptions. Next, a discussion is provided regarding the doses from other pathways. The intent of the discussion is to explore the possibility that there may be other pathways that have the potential for higher normalized collective doses.

5.1 Co-60

Table 9-15 indicates that the total normalized collective dose for Co-60 is about 1E4 person rem/Ci, and it is entirely due to the Co-60 that may find its way into consumer products made from scrap metal. The value was derived via a multi-step process. First, an allocation was assigned to the destination of each unit of activity of Co-60 that is released in the scrap metal. The allocation is as follows:



Note that this allocation has all the Co-60 in the scrap metal partitioning to the melt and then the melt is used to manufacture a wide variety of products, which are also assigned partition fractions.

Independent of this allocation, a set of models were developed to quantify the collective dose per Ci for each box in the above figure. These values are referred to as unweighted collective normalized doses. The above fractions are then multiplied by the unweighted collective normalized doses to obtain the weighted collective normalized doses, which are presented in Table 9-15.

Several cases were run for each unweighted collective normalized dose. These cases were based on the degree to which the product or byproduct into which the radioisotope is distributed is itself recycled after its useful life. For example, after its useful life, an automobile may be scrapped and the steel disposed or recycled. The more limiting case is if the product is assumed to be continually recycled. For the purposes of this section, we investigate the limiting case, where the product or byproduct is assumed to be continually recycled, thereby accumulating collective doses over a 1000 year period.

Table L.5-1 presents the unweighted collective doses for Co-60, along with the weighting factor, and presents how the values in Table 9-15 were derived. The full array of all the possible final destinations of any radionuclide is provided to demonstrate the full scope of the analysis. A discussion of each of these pathways and how they were modeled is provided in Chapter 9.

The table, which provides an overview of how the value of 1E4 person rem/Ci was obtained, is used as the first step in understanding the uncertainties and sensitivities in the derived values. The collective normalized dose and the associated uncertainties for Co-60 are based on the fundamental premise that the Co-60 goes entirely to the melt, and therefore only the doses delivered from commercial products are relevant. Note that, if the Co-60 were postulated to go to the slag, the collective doses would be lower because the unweighted collective doses for the slag pathways are lower.

In order to gain insight into the uncertainties in the normalized collective dose, insight is needed into the uncertainties associated with the derivation of the unweighted collective doses for commercial products and the assigned weighting factors for each product. The most important commercial products are the steel in commercial buildings and the steel in a car.

The unweighted normalized dose for these two different geometries were derived using MicroShield™ and a set of complex assumptions regarding the exposure geometries and settings. Before proceeding with the calculation of these doses, an important concept regarding the derivation of collective doses needs to be appreciated. The normalized collective dose is the same whether the unit of activity is assumed to be distributed in all automobiles or commercial buildings in the U.S., or if it assumed to be all contained in a single representative automobile or commercial building. The reason is, if the radionuclide is assumed to be widely distributed, the concentration of the radionuclide in the steel in the automobile or commercial building is reduced, but the number of exposed people are increased proportionally. Hence, the collective dose per Ci is the same. As such, when deriving the unweighted normalized collective doses for Co-60, it is assumed that the entire unit of activity (e.g., Curie) is contained in a single representative automobile and in a single representative office setting.

Table L.5-1. Overview of Derivation of Normalized Collective Dose for Co-60

Pathway	Unweighted Dose (person rem/Ci in product)	Weighting Factor	Weighted Dose (person rem/Ci free released)
Commercial Products			
Cast iron pan	563	neg	-
SS pan	53.9	neg	-
Commercial building	6850	.474	3247
Kitchen Appl.	5340	.058	310
Whole Auto	18100	.365	6607
Auto shell	27200	.365	NU
Soil	NC	0	-
Air Emission (year 1)	4.66	5E-5	-
Slag			
Concrete road	32.1	neg	-
Road base	1.61	neg	-
RR ballast	11.1	neg	-
Transportation			
Scrap	7.68E-3	1	neg
Steel	5.62E-2	1	neg
Slag	1.07E-3	0	neg
Dust	2.14E-2	0	neg
Baghouse dust	NC	neg	neg
River	7.11	neg	neg
Dispose			
Municipal	neg	NA	-
Haz waste	neg	NA	-
Total			1E4

NA = Not Applicable
NC = Not Calculated
neg = negligible
NU = calculated but not used in the total

Automobile

Envision an automobile where the Co-60 is distributed throughout all the steel in the car. This will create a radiation field inside the car exposing the occupants. The MicroShield™ computer code was used to derive this complex geometry as described in Section 9.5.1. However, this exposure setting can be approximated by assuming the Ci of Co-60 is uniformly contained in a sphere 3 m in diameter and density of air of 0.0023 g/cm^3. The radiation field at the center of the sphere is derived as follows:

D(person rem) = θ(3.8E5 photons/s-cm^2) x 2.4 MeV/photon x .03 cm^2/g x 1.6E-6 erg/MeV x .01 rad-g/erg x 3600 sec/hr x (1500 person hrs/yr)/.1386/yr

D (person rem) = 17055 person rem/Ci of Co-60 in the metal in a car

Where:

θ = $S_v (1-e^{-usR})/u_s$
θ = photon flux at the center of the sphere (3.8E5 photons/cm^2-sec)
S_v = source strength (2619 photons/cm^3-sec)
u_s = linear attenuation coefficient for air (6.9E-5 cm-1)
R = radius of the sphere (150 cm)
E = photon energy per disintegration (2.4 MeV/dis)
T = average annual vehicle occupancy time (1500 person hrs/yr)
u_a' = mass attenuation coefficient (0.03 cm^2/g)

This approximation agrees fairly well with the value of 18,100 derived for the whole car using MicroShield™. The only significant source of uncertainty in this calculation is the occupancy time for the car. The value selected was 2 people per car for about 2 hours per day, 365 days per year, or about 1500 person hours per year. This parameter was considered to be the average occupancy per vehicle in the U.S. The collective dose is directly proportional to this value. Changes in geometries and more sophisticated modeling methods could improve this estimate, but the effect will likely be less than a factor of two.

Commercial Building

Envision that a commercial building is constructed with steel containing 1 Ci of Co-60. This would include the floors and support structure (I-beams, studs, frame). The exposure of the individuals within the structure would be highly variable depending on the individual's proximity to major steel structures. Section 9.5.3 of the main report describes the complex geometries that were modeled for this scenario using MicroShield™. A simplified approximation of this calculation can be made by assuming that the contaminated steel is in the walls, floor, and ceiling of a room 10 m by 10 m x 3 m and occupied by one person. The dose to the occupant could be approximated by treating the 10 m by 10 m areas as two infinite planes containing the 1 Ci of Co-60 and using the dose conversion factors in Federal Guidance Report No. 12, as follows:

$$
\begin{aligned}
D \text{ (person rem)} &= (1 \text{ Ci}/100 \text{ m}^2) \times 3.7\text{E}10 \text{ Bq/Ci} \times 2.35\text{E-}15 \text{ Sv/s m}^2\text{/Bq} \times \\
&\quad 100 \text{ rem/Sv} \times 3600 \text{ sec/hr} \times 2000 \text{ person hr/yr/}.1386\text{/yr} \\
&= 4517 \text{ person rem/Ci}
\end{aligned}
$$

Considering the simplifying assumptions, the results agree fairly well with the value derived using MicroShield™ (i.e., 6850 person rem/Ci). It is unlikely that the value could be higher, except if the average U.S. office worker population density is significantly greater than 1 person per 100 m^2. The Commercial Building Energy Consumption Survey (CBECS) database published by the Energy Information Agency of the Department of Energy reports the following:

All commercial buildings - 1.04 persons per 100 m^2
Office buildings - 2.8 persons per 100 m^2
Lodging - .82 persons per 100 m^2.

The 95 percent confidence intervals for these estimates are about 10%. Hence, the uncertainty in the overall value is small, but the variability among different types of buildings is on the order of a factor of 2 to 3.

Again, changes in the assumed geometries and more sophisticated modeling methods could change this estimate, but the effect will likely be less than a factor of two.

Kitchen Appliance

A person occupying a kitchen is not unlike a person in an office environment with respect to exposure duration and proximity to metal. As such, it is not surprising and reasonable that the unweighted normalized collective dose for the kitchen scenario is similar to that for the office scenario. The assumed number of person hours of exposure per year in a typical kitchen is an important controlling parameter, as is the assumptions regarding geometries and distances. These parameters can have about a factor of 2 to 3 effect on the results.

Other Pathways

There are certainly a myriad of other pathways where individuals could be exposed to Co-60 contained in metal. The above three cases represent settings where large numbers of people could be in relatively close proximity to large amounts of steel for prolonged periods of time. The frying pan scenario is representative of conditions where individuals are exposed to a much smaller source. It also represents a source of internal contamination. However, since it is a relatively small source of exposure, it is not evaluated here.

Weighting Factors

As described in Section 9.5.5 of the main report, the weighting factors were derived based on estimates of the fraction of the radioactivity that ends up in the melt and the fraction of recycled scrap metal that is used in various products on a national level. For Co-60, 100% of the Co-60 contained in released scrap is assumed to partition to the melt, and there is very little uncertainty in this value. There is, however, considerable uncertainty in the distribution of the scrap metal to different types of products. In the analysis, it was assumed that a large portion of the scrap is used in automobiles (36.5%), the construction of commercial buildings (47.4%), and in kitchen appliances (5.8%). In effect, the estimate assumes that about 90 percent of the recycled scrap is used in products that have a very high potential to cause exposures. If, in fact, the percentage of recycled scrap used in such products is smaller, the weighted normalized collective dose would be correspondingly lower. However, the data regarding the use of scrap metal in automobiles is reliable. Hence, the dose can not be lower than about a factor of 2 or 3. It is also unlikely that the dose could be significantly higher for the same reasons.

Assumed Re-recycling Fraction

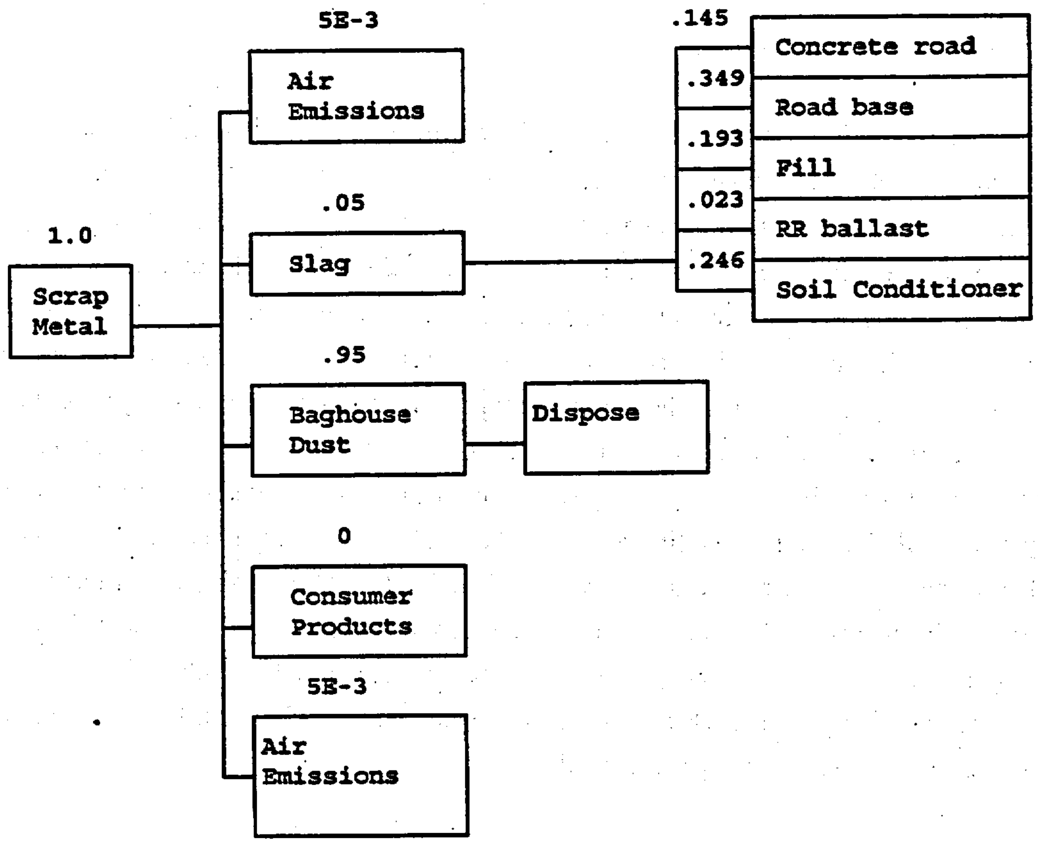
In this discussion, it is assumed that the steel is continually re-recycled, and the doses for a given scenario are accumulated for 1000 years, taking into consideration that the radionuclide inventory is depleting in accordance with its radioactive half life. In reality, the metal product will also have an effective half life because the metal in the automobile, appliance, etc., may eventually be disposed in a municipal landfill, where it poses very little potential for exposure as compared to when the metal is in commercial or domestic use. In the case of Co-60, Table 9-9 of the main report reveals that, whether the steel product is 0% or 100% re-recycled, the time-integrated collective dose remains virtually unchanged. This occurs because the radiological half life for Co-60 is relatively short (i.e., 5 years) as compared to the duration of the first use of the recycled metal. Hence, re-recycling the metal does not significantly increase the time-integrated collective dose. As it turns out, for the radionuclides that partition to the melt, most are relatively short-lived, and the time-integrated collective dose is virtually unaffected by assumptions regarding re-recycling. However, the time-integrated collective doses for the longer-lived radionuclides, such as Tc-99 (2.1E5 years) and Mo-93 (3.5E3 years), increase significantly when re-recycling is assumed.

Conclusions

The combination of the uncertainties in the unweighted normalized collective doses (primarily uncertainties in exposure durations and geometries), together with the uncertainties in the weighting factors (primarily uncertainties in use of recycled scrap), result in an overall modest degree of uncertainty in the normalized collective dose, on the order of a factor of 3 to 5 above to a factor of 3 to 5 below the estimated value of 1E4 person rem/Ci.

5.2 Cs-137

The collective normalized dose for Cs-137 is estimated to be 0.93 person rem/Ci, most of which is attributable to exposure to slag. The allocation is as follows:



This assumed allocation is critical to assessing the normalized collective dose for Cs-137, and its associated uncertainties, for several reasons. Most importantly, 95% of the Cs-137 is assumed to go to baghouse dust, and the dust is assumed to be disposed in a manner that eliminates the potential for exposure. Second, the Cs-137 that does enter the environment via the slag pathway is relatively inaccessible. It is for this reason that the normalized collective dose for Cs-137 is over 4 orders of magnitude lower than that for Co-60; i.e., the Co-60 is assumed to be almost entirely recycled into products to which people can be exposed.

In order to gain insight into the uncertainties in the normalized collective dose for Cs-137, insight is needed into the uncertainties associated with the derivation of the unweighted collective doses for the slag uses and the assigned weighting factors for each use. Table L.5-2 presents an overview of the elements that comprise the normalized collective dose for Cs-137.

Table L.5-2. Overview of Derivation of Normalized Collective Dose for Cs-137

Pathway	Unweighted Dose (person rem/Ci in product)	Weighting Factor	Weighted Dose (person rem/Ci free released)
Commercial Products			
Cast iron pan	NC	NA	-
SS pan	NC	NA	-
Commercial building	NC	NA	-
Kitchen Appl.	NC	NA	-
Whole Auto	NC	NA	-
Auto shell	NC	NA	-
Soil	33.8	.246 x .05	.42
Air Emission (year 1)	3.84	5E-3	.019
Slag			
Concrete road	38.5	.145 x .05	0.28
Road base	0.67	.349 x .05	0.01
RR ballast	29.5	.023 x .05	0.03
Transportation			
Scrap	1.55E-3	1	-
Steel	1.14E-2	0	-
Slag	2.29E-4	.05	-
Dust	4.48E-3	neg	-
Baghouse dust	0	.95	neg
River	41.1	neg	-
Dispose	neg	neg	-
Municipal	neg	NA	-
Haz waste	neg	NA	-
Total			0.76

NA = Not Applicable
NC = Not Calculated
neg = negligible
NU = calculated but not used in the total

The results reveal that the use of slag as a soil conditioner and in concrete in roads are the major contributors to the collective dose because of the relatively high unweighted normalized collective dose and the relatively high weighting factor for these pathways.

Soil Conditioner

The limiting collective dose from Cs-137 in soil is external exposure and vegetable ingestion (EPA 94).

External Exposure:

The time integrated normalized collective dose is derived as follows:

$$
\begin{aligned}
\text{D (person rem)} &= [(1\ \text{Ci} \times .05 \times .246)/.15\ \text{m}^3] \times 1.71\text{E-}17\ \text{Sv/s m}^3/\text{Bq} \times 3.7\text{E}10\ \text{Bq/Ci} \times 100 \\
&\quad \text{rem/Sv} \times 3.15\text{E}7\ \text{s/yr} \times 1\text{E-}4\ \text{persons}/(.696/30\ \text{yr}) \\
&= 0.7\ \text{person rem/Ci}
\end{aligned}
$$

This value agrees well with the computer derived value of 0.76.

In deriving this value, it is assumed that the entire Ci is contained in soil 1 m^2 by 15 cm deep, and that the population density is 1E-4 persons per m^2. The assumed 1 m^2 area is just a convenience. If a larger area were assumed, the concentration would go down but the exposed population would correspondingly increase. Hence, the assumed area has no effect on the results. The assumed population density translates to 100 persons per km^2, which is typical of a suburban area. If the population density were higher, such as 1000 persons per km^2 (as in urbanized areas), the doses would increase by a factor of 10. Conversely, if the population density were assumed to be 10 persons per km^2, as in more rural areas, the dose would decrease by a factor of 10. Since this pathway (i.e., soil conditioning) is associated with agricultural settings, the assumption that the population density is 100 persons/km^2 is a high end estimate.

The analysis also assumes that the activity is uniformly mixed in soil down to a depth of 15 cm. 15 cm was selected as the plow depth for soil conditioning. If the activity were mixed in a thinner layer, such as 1 cm, the normalized dose would increase by about a factor of 2. Other important assumptions imbedded in the analysis is that the exposed population remain in the vicinity of the contaminated soil 100% of the time, and no credit is taken for shielding by structures. If credit were taken for these factors, the dose would be reduced by a factor of 2 to 5.

Vegetable Pathway:

When used as a soil conditioner, the Cs-137 in the slag could be taken up by vegetables grown in the conditioned soil. The associated collective population dose can be estimated as follows:

$$\begin{aligned} \text{D (person rem)} &= [(1\ \text{Ci} \times .05 \times .247)/(1\ \text{m}^2 \times .15\ \text{m} \times 1.6\text{E}6\ \text{kg/m}^3)] \times 1\ \text{kg/yr} \\ &\quad \times (.04\ \text{pCi/kg veg per pCi/kg soil}) \times 1.35\text{E-}8\ \text{Sv/Bq ingested} \\ &\quad \times 3.7\text{E}10\ \text{Bq/Ci} \times 100\ \text{rem/Sv}/(.693/30\text{yr}) \\ &= 4.4\text{E-}3\ \text{person rem/Ci} \end{aligned}$$

The first term in the equation is used to derive the concentration of Cs-137 in soil for a 1 m^2 area and 0.15 m depth. As described above, 1 m^2 is selected for convenience and has no effect on the results. The key sources of uncertainty in the first term are the 0.05 and 0.242 partition factors. If a larger fraction of the Cs-137 were to partition to slag or if a larger fraction of the slag were assumed to be used as a soil conditioner, the doses would be correspondingly higher. Note that, since this is an assessment of collective dose, the fact that there may be some variability in the partitioning at a given mill site is not relevant. The question is, what is the uncertainty in the U.S. aggregate partition factors. The uncertainty in these national aggregate values is likely to be small; i.e., less than a factor of 2.

The 1 kg/yr term in the equation is the quantity of vegetables that is grown per m^2 of agricultural soil. Again, this is a U.S. aggregate term and the uncertainty in this value is likely to be small. The value of 1 kg/m^2-yr is based on a review of Department of Agriculture data which shows this value to be in the mid range. For example, the fresh weight yield reported in EPA 94 ranges from 0.28 kg/m^2 for lima beans to 1.35 kg/m^2 for spinach to 6.98 kg/m^2 for tomatoes.

The next term in the equation, 0.04 pCi/kg of fresh vegetable per pCi/kg of soil, is referred to as the soil-to-plant transfer factor or Bv. This is an empirically determined value. A review of the literature reveals that this is a high end value and is likely to be an overestimate, perhaps by a factor of 10, as applied to the national average. For example, a review of the cesium soil-to-plant transfer factors cited in the literature revealed a geometric mean of 5.0E-3 and a geometric standard deviation of 4.1 (Ng 82)[6].

The remaining terms in the equation can be viewed as constants, with the exception of the denominator, 0.693/30 yr. This term reflects the time period over which the Cs-137 will be in the soil and available for root uptake. It is based on the assumption that the only removal mechanism is radioactive decay. In fact, the Cs-137 will also deplete from the soil by erosion and leaching. This assumption introduces a modest degree of conservatism. For example, assuming an infiltration velocity of 1 m/yr and a K_d of 200, the leaching coefficient for Cs-137 out of the root zone is about 10^{-2}/yr, as compared to the radioactive decay rate of 2E-2/yr; i.e., including leaching in the model would reduce the dose by less than a factor of 2.

The results reveal that the impact from the vegetable pathway is small as compared to the direct radiation pathway for the soil conditioner scenario. This would be the case even if a lower, rural population density were assumed in the derivation of the external exposure dose.

Concrete Road

Slag containing Cs-137 used in concrete for road construction can result in population exposures by the direct radiation exposure of people driving on the roads and of people who live near the roads. The latter pathway is small relative to the drivers because of the distance from the road to the residents. Hence, the analysis is limited to the drivers.

For the purposes of this analysis, let us assume that the 1 Ci of Cs-137 is used in a segment of road 10 m long, 9.14 m wide (30 feet), and 0.24 m thick (9.5 inches), with a density of 2 g/cm³. This results in a concentration of 2.3E4 pCi/g of Cs-137 in the road.

If a person were to stand on that road for an entire year, his or her dose could be approximated as follows:

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[6] Ng, Y.C., et. al., "Soil-to-Plant Concentration Factors for Radiological Assessment," Final report, NUREG/CR-2957, 1982.

$$
\begin{aligned}
D\ (\text{rem/yr}) &= 2.3\text{E}4\ \text{pCi/g} \times 3.76\text{E-}18\ \text{Sv/s m}^3\text{/Bq} \times 100\ \text{rem/Sv} \\
&\quad \times 3.15\text{E}7\ \text{sec/yr} \times 1\text{E}6\ \text{cm}^3\text{/m}^3 \times 2\ \text{g/cm}^3 \times .037\ \text{Bq/pCi} \\
&= 20.2\ \text{rem/yr or } 20.2\ \text{rem per } 8760\ \text{hours per year}
\end{aligned}
$$

The actual exposure setting will be a flow of automobiles over the road. EPA 95 indicates that the flow of people on U.S. roads is 3.88E12 person miles per year, the total road miles is 3,904,721 miles, and the average speed per vehicle is 48,280 m/hr. This results in a flow of 20.56 person hr/m-yr or 205.6 person hrs per 10 m/yr.

Combining these terms, the collective unweighted normalized dose can be approximated as follows:

$$
\begin{aligned}
D\ (\text{person rem}) &= (20.2\ \text{rem per } 8760\ \text{person hr/yr}) \\
&\quad \times (205.6\ \text{person hr/yr})/(0.693/30\ \text{yr}) \\
&= 20.5\ \text{person rem/Ci}
\end{aligned}
$$

This result is about a factor of 2 lower than that derived using MicroShield™.

Other Scenarios

As described above, the normalized collective dose for Cs-137 is extremely small because only 5% of the Cs-137 in the scrap is assumed to partition to the slag. The other 95% partitions to baghouse dust where it is assumed to be disposed of as waste and does not contribute to the collective dose. If the baghouse dust were used, such as in soil conditioner or fill, the collective dose could increase several fold.

Conclusions

The controlling assumptions for the normalized collective dose for Cs-137 are:

1. Only 5% partitions to slag.
2. The 95% that partitions to baghouse dust is disposed of as waste and is inaccessible.

Changes to these assumptions could significantly increase the collective dose.

Alternative modeling assumptions regarding partition fractions to various slag uses, geometries, and occupancy could result in a 2 to 3 fold increase in the normalized collective dose for Cs-137.

6. VARIABILITY AND UNCERTAINTY OF RADIONUCLIDE MINIMUM DETECTABLE CONCENTRATIONS CALCULATED FOR SURFICIALLY- AND VOLUMETRICALLY-CONTAMINATED METALS .

6.1 Introduction

Chapter 8 of the TSD evaluated minimum detectable concentrations (MDCs) of radionuclides in both surface and volume-contaminated metals that may be released for recycling. Several commonly used methods are described which assume optimum survey or counting conditions for each method. As such, the MDCs presented represent the lower end of the MDCs for each method. Under less favorable conditions, the sensitivity could be reduced substantially resulting in MDCs which are higher by about a factor of 2.

The parameters selected for deriving the MDCs presented in Chapter 8 are based largely upon data presented in NUREG-1507 (Huffert 1995), "Minimum Detectable Concentrations with Typical Radiation Survey Instruments for Various Contaminants and Field Conditions." This recently-published report provides an excellent quantitative evaluation of factors affecting the detection sensitivity of commercially available portable field instruments being used to conduct radiological surveys in support of decommissioning.

In general, the parameter values have been selected to calculate optimal MDCs, representing ideal laboratory conditions of low background, smooth, clean, flat surfaces, and experienced survey personnel. MDCs in the field will likely be higher due to such factors as increased background, variable source to detector distance, surface roughness and composition, and the presence of surface coatings such as dust, paint, oil, or water. This is particularly true for alpha emitters, and to some extent for beta-emitting radionuclides. Thus, radionuclides for which the MDC in this analysis is found to be only marginally less than the appropriate guideline level will likely not be detectable in the field.

While volumetric MDCs may be calculated from a knowledge of appropriate parameter values, the MDCs in the TSD for laboratory analysis of solid samples were taken from an article by F.M. Cox and C.F. Guenther (Cox 1995). The authors present a range of MDCs as reported by 24 commercial and government laboratories. The article presents state-of-the-art MDCs and associated parameter values for laboratory analysis of radioactive materials in solids.

The purpose of this report is to discuss the variability that is likely for MDCs for standard monitoring methods under conditions typically encountered during decommissioning surveys.

6.2 Determination of Minimum Detectable Concentrations

Before discussing variability in MDCs, it is important to recall the basic mathematical relationships used to calculate MDC for the various survey methodologies.

6.2.1 Surficial Contamination

The minimum detectable concentration (MDC) of a sample is an *a priori* level of radioactivity that is practically achievable by an overall measurement process (EPA 1980). An excellent discussion of fundamental MDC concepts and measurement methods applicable to monitoring of surface-contaminated materials under field conditions is presented in NUREG-1507.

Surface Scanning for Small Areas of Contamination

The MDC for detection of small areas of contamination using surface scanning is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{W}{60 * V}}}{\frac{W}{60 * V} * (\sum Y_i * \varepsilon_i) * \frac{A}{100} * HF} \qquad (1)$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
W = detector width (cm)
60 = conversion factor (s/min)
V = detector scan rate (cm/s)
Y_i = yield for emission i (ptcle-emitted/d)

ϵ_i = detector efficiency for emission i (c/ptcle-emitted)
A = detector area (cm^2)
HF = surveyor efficiency (%).

Surface Scanning for Large Areas of Contamination

The MDC for detection of large areas of contamination using surface scanning is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{2 * \tau}{60}}}{\frac{2 * \tau}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100} * HF} \tag{2}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
t = meter time constant (s)
60 = conversion factor (s/min)
Y_i = yield for emission i (ptcle-emitted/d)
ϵ_i = detector efficiency for emission i (c/ptcle-emitted)
A = detector area (cm^2)
HF = surveyor efficiency (%).

Direct Measurements

The MDC for direct measurements is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * \frac{t}{60}}}{\frac{t}{60} * (\sum Y_i * \varepsilon_i) * \frac{A}{100}} \tag{3}$$

where:

MDC = minimum detectable concentration (dpm/100cm^2)
B_R = detector background count rate (cpm)
t = count time (s)
60 = conversion factor (s/min)
Y_i = yield for emission i (ptcle-emitted/d)
ϵ_i = detector efficiency for emission i (c/ptcle-emitted)
A = detector area (cm^2).

6.2.2 Volumetric Contamination

The MDC for laboratory measurements is calculated using the following equation:

$$MDC = \frac{3 + 4.65\sqrt{B_R * t}}{t * Y_i * \varepsilon_i * M * R} \qquad (4)$$

where:

MDC = minimum detectable concentration (pCi/g)
B_R = detector background count rate (cpm)
t = count time (s)
Y_i = yield for emission i (ptcle/d)
ϵ_i = detector efficiency for emission i (c/ptcle)
M = sample mass (g)
R = chemical yield.

6.3 Variability of Minimum Detectable Concentrations

The variability of MDC values may be viewed through a discussion of the variability of the parameter values that may be used to compute MDC. In general, the parameters include variables that determine the instrument MDC and variables that can affect the counting efficiency of the instrument in the field. The following is a discussion of the variability of each of the parameters that are used in MDC calculations.

6.3.1 Background Count Rate

Background count rates, B_R, presented in the TSD have been taken from NUREG-1507. Higher background count rates will result in increased MDCs. Examination of equations (1) - (3) show that the MDC will increase as the square root of the increase in the background count rate. Background levels will increase in the field, particularly at locations near nuclear facilities, where surveys for release of metals for recycle will be performed. For beta counting, the pancake GM detector will show a very slight increase with increasing background, 3.6 cpm per μR/h, while the gas proportional counter will show an increase of 25 cpm per μR/h. Gamma counting with a high sensitivity 2 x 2-inch NaI scintillator will be significantly affected by increasing background, with increases of 1200 cpm per μR/h. While typical environmental exposure rates may be in the range of 6 - 10 μR/h, ambient "background" levels at a nuclear facility could easily range from 50 to 100 μR/h or higher and vary considerably over time depending upon current decommissioning activities.

6.3.2 Detector Dimensions

The detector width, W, and area, A, values are specific to the detectors used in the MDC calculations. Use of detectors other than those used in the MDC calculations will not only change these two parameters but will also affect background count rate and detector efficiency. However, all other factors being constant, using a detector with a smaller area will result in a direct increase in MDC. Using a detector with a different width than that used in the calculations is discussed further in the next section.

6.3.3 Detector Scan Rate

For the calculation of alpha and beta MDCs for small areas of contamination using surface scanning, the value of the detector scan rate, V, has been chosen to correspond to a rate of one-third detector width per second. This value is considered to be the slowest scan rate acceptable to the industry and thus results in the lowest MDCs for detection of small areas of contamination. (A faster scan rate of one detector width per second is more commonly used in the industry, but with a higher MDC.) In surveying for small areas of elevated activity, a scan rate of one-third detector width per second will result in an "observation interval" of 3 seconds. The observation interval is the time that the detector is positioned over a small area of elevated activity. For this analysis, the observation interval is assumed to be given by the ratio of the detector width and the detector scan rate (W/V). A decrease in observation interval, such as through use of a faster scan rate, will result in an increase in MDC, proportional to the square root of the reduction in the observation interval.

For gamma scanning MDC calculations, a faster scan rate has been used in keeping with common field practice. Scan rates of 50 cm/s and 15 cm/s have been used for monitoring large and small areas of contamination.

6.3.4 Ratemeter Time Constant

For the calculation of MDCs for large areas of contamination using surface scanning, the value selected for the meter time constant, τ (10 seconds), is the slow response setting on standard ratemeters. Use of the meter fast response (a time constant of 2 seconds) will result in MDCs that are about a factor of 2.2 lower (proportional to the square root of the reduction in the time constant).

6.3.5 Count Time

For the calculation of MDCs for direct measurements, a count time, t, of 1 minute was used. This value is totally arbitrary, but is considered to be a reasonable one, capable of producing relatively low MDCs. Higher or lower count times can be selected as dictated by the situation to reduce survey time (if a higher MDC would be acceptable) or to achieve a lower MDC (with a resulting increase in survey duration). Counting for a shorter time will result in higher MDCs. As noted above for the observation interval, the increase will be proportional to the square root of the reduction in the count time.

6.3.6 Human Factors Efficiency

The human factors efficiency, HF, used in the calculation of the MDC for scanning attempts to account for the fact that the probability of detecting residual contamination in the field is not only affected by the sensitivity of the survey instrumentation when used in the scanning mode of operation, but also by the surveyor's ability. The surveyor must decide whether the signals represent only the background activity, or whether they represent residual contamination in excess of background. The values selected for this parameter are based upon empirical results presented in NUREG-1507. There is a fair degree of variability present in the empirical data for this parameter. This is due in part to subjective matters affecting the surveyor's decision-making, including the relative costs of "misses" and "false positives," and the surveyor's assumptions regarding the likelihood of contamination being present.

A HF value of 60% was used for the GM detector, while 80% was used for the GP detector. In NUREG-1507, this parameter ranges from 40 - 70% for the GM and from 65 - 90% for the GP.

6.3.7 Counting Efficiency

The overall detector efficiency used to calculate MDCs for a radionuclide is given by the summation over all emissions of each type particle of the product of the yield of each particle emitted in the decay of the radionuclide, Y_i, and the efficiency of the particular detector for that particle type and energy, e_i. Particle yields and energies for each radionuclide have been taken from the NUCDECAY data library produced by the Dosimetry Research Group of the Health Sciences Research Division at Oak Ridge National Laboratory (Eckerman 1993). These values are derived from the Evaluated Nuclear Structure Data File (ENSDF) at the Brookhaven National Laboratory. None of the ENSDF data is newer than 1979. Any significant changes in decay data since that time are therefore not incorporated in these MDC results.

There are a number of factors that may potentially affect detector efficiency. Detector efficiency may be considered to be composed of two components, instrument efficiency and source efficiency. The product of the instrument efficiency and source efficiency yields the total efficiency. The instrument efficiency is the ratio of the net count rate of the instrument and the surface emission rate of a source for a specified geometry. The source efficiency is the ratio of the number of particles of a given type emerging from the front face of a source and the number of particles of the same type created or released within the source per unit time. The source efficiency takes into account the increased particle emission due to backscatter effects, as well as the decreased particle emission due to self-absorption losses.

Surface contamination is assessed by converting the instrument response to surface activity using one overall total efficiency. This is not a problem provided that the calibration source exhibits similar characteristics as does the surface contamination (e.g., radiation energy, backscatter effects, source geometry, self-absorption, etc.). In practice, instrument efficiencies are determined with a clean, smooth, stainless steel source in a laboratory, and then those efficiencies are used to measure contamination on a multitude of surface conditions in the field.

The efficiency of GP and GM detectors for beta particles is a strong function of particle energy. With decreasing energy, an increasing fraction of the betas reaching the detector fail to penetrate the window, resulting in a reduced efficiency. The detector efficiencies have been derived from data published for several radionuclides and detectors of NUREG-1507.

When surveying for alpha-emitters, those alpha particles reaching the detector are generally of sufficient energy to penetrate the window and result in a count. Therefore, a constant detector efficiency was assumed for the GP detector and for the zinc sulfide scintillator for all alpha-emitters, based upon data in NUREG-1507.

For gamma emitters, the overall counting efficiency is dependent upon the source-detector geometry. Cases have been run for both a small, localized area of contamination and a relatively large area. In both cases, the detector was assumed to be 6 cm away from the source.

The efficiencies used in the calculation of MDCs are for ideal laboratory conditions, which include the use of clean, smooth, calibration sources, under a controlled source-detector geometry. In the field, the distance between the source and the detector may be different than that used in calibration. Instruments are typically calibrated at a distance of ½ to 1 cm from the detector. If used at distances that are greater than the calibration distance, the actual level of activity could be greater than assumed. The greatest reduction in detector response (and resulting increase in MDC) per increased distance from the source would be observed for alpha emitters and low energy beta emitters such as Ni-63 and C-14. Increasing the distance from ½ cm to 2 cm causes an efficiency reduction ranging from a factor of 2 for C-14, 4 for Pu-239, and 10 for Ni-63. For higher energy beta emitters such as Sr/Y-90, the reduction is only 25%.

In certain field situations such as monitoring pieces of equipment with non-flat surfaces (i.e., interior of pipe), the surveyor may be unable to place the detector within 1 cm of the surface. A lower efficiency should be assumed in such cases.

The source-detector geometry is another factor that may cause a degree of variability in calculated MDCs. The MDC calculations presented in the TSD used efficiencies determined for distributed sources. However, the detector's response may be influenced by the contaminant's distribution on the surface being monitored. NUREG-1507 identifies a 6 to 42% greater efficiency for disc sources (5 cm^2) than those obtained for distributed sources. This is expected because of the solid angle of the measurement geometry. For the smaller disc source, a larger fraction of the radiation particles emitted from the source intersect the probe area. Use of distributed source efficiencies in the MDC calculation results in an underestimate of the actual MDC if the source is likely to be characterized by small localized areas of contamination.

The source efficiency will be impacted by field surface types and conditions, particularly by those that may affect the usefulness of a particular instrument. One of the more significant implicit assumptions made during instrument calibration and subsequent use of the instrument in the field is that the composition and geometry of contamination in the field is the same as that of the calibration source. This may not be the case, considering that many calibration sources are fabricated from materials different from those that comprise the surfaces of interest in the field. This difference usually manifests itself in the varying backscatter characteristics of the calibration and field surface materials.

The effects of surface condition on detection sensitivity have been evaluated in NUREG-1507. The conversion of the surface emission rate to the activity of the contamination source is often a complicated task that may result in significant uncertainty if there are deviations from the assumed source geometry. The data in NUREG-1507 for source efficiencies for several common surface types indicate that the source efficiency varies widely depending upon the amount of self-absorption and backscatter provided by the surface. Based upon this data, source efficiencies for smooth steel surfaces will be relatively high due to appreciable backscatter. However, if the metal surface has been severely abraded or has become pitted due to decontamination, the source efficiencies may be relatively lower due to self-absorption effects. The overall impact will be dependent upon the surface of the source used for calibration, which is typically a clean, high-backscatter reference source and that of the materials being released for recycle.

Another significant source of variability in counting efficiency is due to attenuation effects of overlying material. NUREG-1507 has evaluated the effects of known thicknesses of paint, water, dust, and oil. The effects of 1.5 to 4.5 mg/cm^2 of oil, 1.9 to 12.6 mg/cm^2 of paint, 2.3 to 10.0 mg/cm^2 of dust, and 0.44 to 7.6 mg/cm^2 of water were evaluated. One interesting finding was that the total density thickness produced approximately the same relative amount of attenuation, regardless of the material responsible for the attenuation. For beta radiation, a 10 mg/cm^2 thickness of absorber only causes a 10 - 40% reduction in overall efficiency. However, for alpha radiation, a density thickness of 6 mg/cm^2 is sufficient to attenuate almost all of the alpha particles.

6.3.8 Laboratory MDCs

The MDCs for laboratory analysis of volumetric contaminants were based upon a survey of commercial and government laboratories which provided their "best estimates" in response to the survey. The instrumentation used, instrument efficiencies, background count rates, count times, and sample masses varied from one radionuclide to another. The MDC values presented in the TSD represent the lowest reported in the survey. However, the survey results are presented as a range of MDCs. Variations among the different laboratories in the reported MDCs can be traced back to variabilities of each laboratory in terms of sample sizes utilized, count times, operational detection efficiencies, and typical background levels. Lower MDCs can be obtained in some cases through use of increased counting times or an increase in sample mass. While increasing the count time or sample mass can lead to the detection of lower concentrations, it should be realized that there are practical limits on detection which are driven by time constraints, background levels, and the desired level of confidence. It should also be noted that the overall time it takes to process a sample is dependent upon the number of isotopes to be analyzed, the amount of sample preparation required, the availability of a laboratory that can handle the samples, and the time it takes to collect, deliver, and retrieve the sample results.

In general, with reasonable counting times, i.e., 10 to 1,000 minutes, MDCs are in the range of 0.1 to 1.0 pCi/g for nearly all isotopes. Detection at levels below 0.1 pCi/g, while feasible, is not economical for commercial laboratories. This in turn leads to the conclusion that while longer counting times drive MDCs lower, the costs associated with tying up the detector for a longer period of time were not necessarily economical.

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